3.0 METHODS TO CONTROL NUTRIENTS

3.1 INTRODUCTION

One of the most effective ways to control algal populations is by limiting the nutrient supply to the lake, and thus limiting growth of algae. This approach may work with some rooted aquatic plants as well, but as most rooted plants acquire most of their nutrition from the sediment (Barko and Smart, 1981), control of nutrients in the water column is far more effective as an algal management strategy. As previously discussed in Section 1, phosphorus is the best nutrient to control, and the discussion of this section will deal primarily with phosphorus control methods. In nutrient rich lakes, the growth of algae may be limited by light, and reduction in nutrient concentrations may not have a significant effect until the nutrient concentrations are lowered sufficiently to induce nutrient limitation (Section 1).

One must usually identify the sources of nutrients before an effective control strategy can be determined. To do this, an accurate phosphorus budget is required (Section 1.2). Once the relative importance of the sources of phosphorus is determined, one can examine the control techniques identified below for applicability and feasibility, and compare them to the "No Management Alternative" for nutrients.

3.1 Non-Point Sources – control of diffuse nutrient sources from the watershed
3.2 Point Sources – control of point sources, usually piped discharges
3.3 Hydraulic Controls – diversion, dilution, flushing, and hypolimnetic withdrawal strategies
3.4 Phosphorus Inactivation – chemical binding of phosphorus to limit availability
3.5 Artificial Circulation and Aeration – mixing and oxygen addition
3.6 Dredging – removal of nutrient-laden sediments
3.7 Additional Techniques – bacterial additives and removal of bottom feeding fish

The expected reduction in phosphorus loading should be modeled as described in Section 1 to predict the change in trophic status. In general, algal problems will be minimized at loadings less than Vollenweider's permissible level, but algal abundance in response to nutrient loading is a probability distribution, not a threshold function. Consequently, algal blooms may be expected at some reduced frequency, even at fairly low nutrient levels, and lakes will not respond identically to changes in loading. Acceptable results might be achieved at loadings higher than the permissible level, but unacceptable conditions can be expected where loading exceeds Vollenweider's critical limit. Managers should be prepared to adjust strategies in response to resultant lake conditions; algal control through nutrient limitation is often an iterative process.

Additional ways to directly limit the density of algae may be needed on an interim or supplemental basis, and include the use of biocidal chemicals, dyes or biocontrol agents. These are addressed in Section 4. A summary table of available techniques is presented in Table 3-1, adapted from Wagner (2001). Techniques that address nutrient levels are mixed with methods aimed more directly at the algae, providing a complete summary of
algae control approaches, including the nutrient controls that are the subject of this section

Table 3-1 Options for control of algae. (Adapted from Wagner 2001).

OPTION	MODE OF ACTION	ADVANTAGES	DISADVANTAGES
Watershed controls 1) Management for nutrient input reduction	♦ Includes wide range of watershed and lake edge activities intended to eliminate nutrient sources or reduce delivery to lake ♦ Essential component of algal control strategy where internal recycling is not the dominant nutrient source, and desired even where internal recycling is important	♦ Acts against the original source of algal nutrition ♦ Creates sustainable limitation on algal growth ♦ May control delivery of other unwanted pollutants to lake ♦ Facilitates ecosystem management approach which considers more than just algal control	♦ May involve considerable lag time before improvement observed ♦ May not be sufficient to achieve goals without some form of in-lake management ♦ Reduction of overall system fertility may impact fisheries ♦ May cause shift in nutrient ratios which favor less desirable algae
1a) Point source controls	♦ More stringent discharge requirements ♦ May involve diversion ♦ May involve technological or operational adjustments ♦ May involve pollution prevention plans	♦ Often provides major input reduction ♦ Highly efficient approach in most cases ♦ Success easily monitored	♦ May be very expensive in terms of capital and operational costs ♦ May transfer problems to another watershed ♦ Variability in results may be high in some cases
1b) Non-point source controls	♦ Reduction of sources of nutrients ♦ May involve elimination of land uses or activities that release nutrients ♦ May involve alternative product use, as with no phosphate fertilizer	♦ Removes source ♦ Limited or no ongoing costs	♦ May require purchase of land or activity ♦ May be viewed as limitation of “quality of life” ♦ Usually requires education and gradual implementation

Table 3 - continued

Option	Mode of Action	ADVANTAGES	DISADVANTAGES
1c) Non-point source pollutant trapping	♦ Capture of pollutants between source and lake ♦ May involve drainage system alteration ♦ Often involves wetland treatments (detention/infiltration) ♦ May involve stormwater collection and treatment as with point sources	♦ Minimizes interference with land uses and activities ♦ Allows diffuse and phased implementation throughout watershed ♦ Highly flexible approach ♦ Tends to address wide range of pollutant loads	♦ Does not address actual sources ♦ May be expensive on necessary scale ♦ May require substantial maintenance
IN-LAKE PHYSICAL CONTROLS			
2) Circulation and destratification	♦ Use of water or air to keep water in motion ♦ Intended to prevent or break stratification ♦ Generally driven by mechanical or pneumatic force	♦ Reduces surface build-up of algal scums ♦ May disrupt growth of blue-green algae ♦ Counteraction of anoxia improves habitat for fish/invertebrates ♦ May reduce internal loading of phosphorus	♦ May spread localized impacts ♦ May lower oxygen levels in shallow water ♦ May promote downstream impacts
3) Dilution and flushing	♦ Addition of water of better quality can dilute nutrients ♦ Addition of water of similar or poorer quality flushes system to minimize algal build-up ♦ May have continuous or periodic additions	♦ Dilution reduces nutrient concentrations without altering load ♦ Flushing minimizes detention; response to pollutants may be reduced	♦ Diverts water from other uses ♦ Flushing may wash desirable zooplankton from lake ♦ Use of poorer quality water increases loads ♦ Possible downstream impacts
4) Drawdown	♦ Lowering of water over autumn period allows oxidation, desiccation and compaction of sediments ♦ Duration of exposure and degree of dewatering of exposed areas are important ♦ Algae are affected mainly by reduction in available nutrients.	♦ May reduce available nutrients or nutrient ratios, affecting algal biomass and composition ♦ Opportunity for shoreline clean-up/structure repair ♦ Flood control utility ♦ May provide rooted plant control as well	♦ Possible impacts on non-target resources ♦ Possible impairment of water supply ♦ Alteration of downstream flows and winter water level ♦ May result in greater nutrient availability if flushing inadequate

Table 3 - continued

Option	Mode of Action	ADVANTAGES	DISADVANTAGES
5) Dredging	♦ Sediment is physically removed by wet or dry excavation, with deposition in a containment area for dewatering ♦ Dredging can be applied on a limited basis, but is most often a major restructuring of a severely impacted system ♦ Nutrient reserves are removed and algal growth can be limited by nutrient availability	♦ Can control algae if internal recycling is main nutrient source ♦ Increases water depth ♦ Can reduce pollutant reserves ♦ Can reduce sediment oxygen demand ♦ Can improve spawning habitat for many fish species ♦ Allows complete renovation of aquatic ecosystem	♦ Temporarily removes benthic invertebrates ♦ May create turbidity ♦ May eliminate fish community (complete dry dredging only) ♦ Possible impacts from containment area discharge ♦ Possible impacts from dredged material disposal ♦ Interference with recreation or other uses during dredging
5a) "Dry" excavation	♦ Lake drained or lowered to maximum extent practical ♦ Target material dried to maximum extent possible ♦ Conventional excavation equipment used to remove sediments	♦ Tends to facilitate a very thorough effort ♦ May allow drying of sediments prior to removal ♦ Allows use of less specialized equipment	♦ Rarely truly a dry operation; tends to be messy ♦ Eliminates most aquatic biota unless a portion left undrained ♦ Eliminates lake use during dredging
5b) "Wet" excavation	♦ Lake level may be lowered, but sediments not substantially exposed ♦ Draglines, bucket dredges, or long-reach backhoes used to remove sediment	♦ Requires least preparation time or effort, tends to be least cost dredging approach ♦ May allow use of easily acquired equipment ♦ May preserve aquatic biota	♦ Usually creates extreme turbidity ♦ Normally requires intermediate containment area to dry sediments prior to hauling ♦ May disrupt ecological function ♦ Disrupts many uses
5c) Hydraulic removal	♦ Lake level not reduced ♦ Suction or cutterhead dredges create slurry which is hydraulically pumped to containment area ♦ Slurry is dewatered; sediment retained, water discharged	♦ Creates minimal turbidity and impact on biota ♦ Can allow some lake uses during dredging ♦ Allows removal with limited access or shoreline disturbance	♦ Often leaves some sediment behind ♦ Cannot handle coarse or debris-laden materials ♦ Requires sophisticated and more expensive containment area

Table 3 - continued

Option	Mode of Action	ADVANTAGES	DISADVANTAGES
6) Light-limiting dyes and surface covers	♦ Creates light limitation	♦ Creates light limit on algal growth without high turbidity or great depth ♦ May achieve some control of rooted plants as well	♦ May cause thermal stratification in shallow ponds ♦ May facilitate anoxia at sediment interface with water
6.a) Dyes	♦ Water-soluble dye is mixed with lake water, thereby limiting light penetration and inhibiting algal growth ♦ Dyes remain in solution until washed out of system.	♦ Produces appealing color ♦ Creates illusion of greater depth	♦ May not control surface bloom-forming species ♦ May not control growth of shallow water algal mats ♦ Alters thermal regime
6.b) Surface covers	♦ Opaque sheet material applied to water surface	♦ Minimizes atmospheric and wildlife pollutant inputs	♦ Minimizes atmospheric gas exchange ♦ Limits recreational use
7) Mechanical removal	♦ Filtering of pumped water for water supply purposes ♦ Collection of floating scums or mats with booms, nets, or other devices ♦ Continuous or multiple applications per year usually needed	♦ Algae and associated nutrients can be removed from system ♦ Surface collection can be applied as needed ♦ May remove floating debris ♦ Collected algae dry to minimal volume	♦ Filtration requires high backwash and sludge handling capability for use with high algal densities ♦ Labor and/or capital intensive ♦ Variable collection efficiency ♦ Possible impacts on non-target aquatic life
8) Selective withdrawal	♦ Discharge of bottom water which may contain (or be susceptible to) low oxygen and higher nutrient levels ♦ May be pumped or utilize passive head differential	♦ Removes targeted water from lake efficiently ♦ Complements other techniques such as drawdown or aeration ♦ May prevent anoxia and phosphorus build up in bottom water ♦ May remove initial phase of algal blooms which start in deep water ♦ May create coldwater conditions downstream	♦ Possible downstream impacts of poor water quality ♦ May eliminate colder thermal layer that supports certain fish ♦ May promote mixing of remaining poor quality bottom water with surface waters ♦ May cause unintended drawdown if inflows do not match withdrawal

Table 3 - continued

Option	Mode of Action	ADVANTAGES	DISADVANTAGES
9) Sonication	♦ Sound waves disrupt algal cells	♦ Supposedly affects only algae (new technique) ♦ Applicable in localized areas	♦ Uncertain effects on non-target organisms ♦ May release cellular toxins or other undesirable contents into water column
IN-LAKE Chemical controls			
10) Hypolimnetic aeration or oxygenation	♦ Addition of air or oxygen at varying depth provides oxic conditions ♦ May maintain or break stratification ♦ Can also withdraw water, oxygenate, then replace	♦ Oxic conditions promote binding/sedimentation of phosphorus ♦ Counteraction of anoxia improves habitat for fish/invertebrates ♦ Build-up of dissolved iron, manganese, sulfide, ammonia and phosphorus reduced	♦ May accidentally disrupt thermal layers important to fish community ♦ Theoretically promotes supersaturation with gases harmful to fish ♦ Biota may become dependent on continued aeration
11) Algaecides	♦ Liquid or pelletized algaecides applied to target area ♦ Algae killed by direct toxicity or metabolic interference ♦ Typically requires application at least once/yr, often more frequently	♦ Rapid elimination of algae from water column, normally with increased water clarity ♦ May result in net movement of nutrients to bottom of lake	♦ Possible toxicity to non-target species ♦ Restrictions on water use for varying time after treatment ♦ Increased oxygen demand and possible toxicity ♦ Possible recycling of nutrients
11a) Forms of copper	♦ Cellular toxicant, suggested disruption of photosynthesis, nitrogen metabolism, and membrane transport ♦ Applied as wide variety of liquid or granular formulations, often in conjunction with chelators, polymers, surfactants or herbicides	♦ Effective and rapid control of many algae species ♦ Approved for use in most water supplies	♦ Possible toxicity to aquatic fauna ♦ Ineffective at colder temperatures ♦ Accumulation of copper in system ♦ Resistance by certain green and blue-green nuisance species ♦ Rupturing of cells releases nutrients and toxins

Table 3 - continued

Option	Mode of Action	ADVANTAGES	DISADVANTAGES
11b) Synthetic organic herbicides	♦ Absorbed or membrane-active chemicals which disrupt metabolism ♦ Causes structural deterioration	♦ Used where copper is ineffective ♦ Limited toxicity to fish at recommended dosages ♦ Rapid action	♦ Non-selective in treated area ♦ Possible toxicity to aquatic fauna (varying degrees by dose and formulation) ♦ Time delays on water use
11c) Oxidants	♦ Disrupts most cellular functions, tends to attack membranes ♦ Applied most often as a liquid.	♦ Potential selectivity against blue-greens ♦ Moderate control of thick algal mats, used where copper alone is ineffective ♦ Rapid action	♦ Older formulations tended to have high toxicity to some aquatic fauna ♦ New formulations not well tested in the field yet
12) Phosphorus inactivation	♦ Typically salts of aluminum, iron or calcium are added to the lake, as liquid or powder ♦ Phosphorus in the treated water column is complexed and settled to the bottom of the lake ♦ Phosphorus in upper sediment layer is complexed, reducing release from sediment ♦ Permanence of binding varies by binder in relation to redox potential and pH	♦ Can provide rapid, major decrease in phosphorus concentration in water column ♦ Can minimize release of phosphorus from sediment ♦ May remove other nutrients and contaminants as well as phosphorus ♦ Flexible with regard to depth of application and speed of improvement	♦ Possible toxicity to fish and invertebrates, mainly by aluminum at low or high pH ♦ Possible release of phosphorus under anoxia (with Fe) or extreme pH (with Ca) ♦ May cause fluctuations in water chemistry, especially pH, during treatment ♦ Possible resuspension of floc in shallow areas ♦ Adds to bottom sediment, but typically an insignificant amount
13) Sediment oxidation	♦ Addition of oxidants, binders and pH adjusters to oxidize sediment ♦ Binding of phosphorus is enhanced ♦ Denitrification is stimulated	♦ Can reduce phosphorus supply to algae ♦ Can alter N:P ratios in water column ♦ May decrease sediment oxygen demand	♦ Possible impacts on benthic biota ♦ Longevity of effects not well known ♦ Possible source of nitrogen for blue-green algae

Table 3 - continued

Option	Mode of Action	ADVANTAGES	DISADVANTAGES
14) Settling agents	♦ Closely aligned with phosphorus inactivation, but can be used to reduce algae directly too ♦ Lime, alum or polymers applied, usually as a liquid or slurry ♦ Creates a floc with algae and other suspended particles ♦ Floc settles to bottom of lake ♦ Re-application typically necessary at least once/yr	♦ Removes algae and increases water clarity without lysing most cells ♦ Reduces nutrient recycling if floc sufficient ♦ Removes non-algal particles as well as algae ♦ May reduce dissolved phosphorus levels at the same time	♦ Possible impacts on aquatic fauna ♦ Possible fluctuations in water chemistry during treatment ♦ Resuspension of floc possible in shallow, well-mixed waters ♦ Promotes increased sediment accumulation
15) Selective nutrient addition	♦ Ratio of nutrients changed by additions of selected nutrients ♦ Addition of non-limiting nutrients can change composition of algal community ♦ Processes such as settling and grazing can then reduce algal biomass (productivity can actually increase, but standing crop can decline)	♦ Can reduce algal levels where control of limiting nutrient not feasible ♦ Can promote non-nuisance forms of algae ♦ Can improve productivity of system without increased standing crop of algae	♦ May result in greater algal abundance through uncertain biological response ♦ May require frequent application to maintain desired ratios ♦ Possible downstream effects
IN-LAKE BIOLOGICAL CONTROLS			
16) Enhanced grazing	♦ Manipulation of biological components of system to achieve grazing control over algae ♦ Typically involves alteration of fish community to promote growth of large herbivorous zooplankton, or stocking with phytophagous fish	♦ May increase water clarity by changes in algal biomass or cell size distribution without reduction of nutrient levels ♦ Can convert unwanted biomass into desirable form (fish) ♦ Harnesses natural processes to produce desired conditions	♦ May involve introduction of exotic species ♦ Effects may not be controllable or lasting ♦ May foster shifts in algal composition to even less desirable forms

Table 3 - continued

Option	Mode of Action	ADVANTAGES	DISADVANTAGES
16.a) Herbivorous fish (not permitted in MA)	♦ Stocking of fish that eat algae	♦ Converts algae directly into potentially harvestable fish ♦ Grazing pressure can be adjusted through stocking rate	♦ Typically requires introduction of non-native species ♦ Difficult to control over long term ♦ Smaller algal forms may be benefited and bloom
16.b) Herbivorous zooplankton	♦ Reduction in planktivorous fish to promote grazing pressure by zooplankton ♦ May involve stocking piscivores or removing planktivores ♦ May also involve stocking zooplankton or establishing refugia	♦ Converts algae indirectly into harvestable fish ♦ Zooplankton response to increasing algae can be rapid ♦ May be accomplished without introduction of non-native species ♦ Generally compatible with most fishery management goals	♦ Highly variable response expected; temporal and spatial variability may be high ♦ Requires careful monitoring and management action on 1-5 yr basis ♦ Larger or toxic algal forms may be benefited and bloom
17) Bottom-feeding fish removal	♦ Removes fish that browse among bottom deposits, releasing nutrients to the water column by physical agitation and excretion	♦ Reduces turbidity and nutrient additions from this source ♦ May restructure fish community in more desirable manner	♦ Targeted fish species are difficult to eradicate or control ♦ Reduction in fish populations valued by some lake users (human/non-human)
18) Pathogens	♦ Addition of inoculum to initiate attack on algal cells ♦ May involve fungi, bacteria or viruses	♦ May create lakewide "epidemic" and reduction of algal biomass ♦ May provide sustained control through cycles ♦ Can be highly specific to algal group or genera	♦ Largely experimental approach at this time ♦ May promote resistant nuisance forms ♦ May cause high oxygen demand or release of toxins by lysed algal cells ♦ Effects on non-target organisms uncertain
19) Competition and allelopathy	♦ Plants may tie up sufficient nutrients to limit algal growth ♦ Plants may create a light limitation on algal growth ♦ Chemical inhibition of algae may occur through substances released by other organisms	♦ Harnesses power of natural biological interactions ♦ May provide responsive and prolonged control	♦ Some algal forms appear resistant ♦ Use of plants may lead to problems with vascular plants ♦ Use of plant material may cause depression of oxygen levels

Table 3 - continued

Option	Mode of Action	ADVANTAGES	DISADVANTAGES
19a) Plantings for nutrient control	♦ Plant growths of sufficient density may limit algal access to nutrients ♦ Plants can exude allelopathic substances which inhibit algal growth ♦ Portable plant "pods" , floating islands, or other structures can be installed	♦ Productivity and associated habitat value can remain high without algal blooms ♦ Can be managed to limit interference with recreation and provide habitat ♦ Wetland cells in or adjacent to the lake can minimize nutrient inputs	♦ Vascular plants may achieve nuisance densities ♦ Vascular plant senescence may release nutrients and cause algal blooms ♦ The switch from algae to vascular plant domination of a lake may cause unexpected or undesirable changes
19b) Plantings for light control	♦ Plant species with floating leaves can shade out many algal growths at elevated densities	♦ Vascular plants can be more easily harvested than most algae ♦ Many floating species provide valuable waterfowl food	♦ At the necessary density, floating plants likely to be a recreational nuisance ♦ Low surface mixing and atmospheric contact promote anoxia
19c) Addition of barley straw	♦ Input of barely straw can set off a series of chemical reactions which limit algal growth ♦ Release of allelopathic chemicals can kill algae ♦ Release of humic substances may bind phosphorus	♦ Materials and application are relatively inexpensive ♦ Decline in algal abundance is more gradual than with algaecides, limiting oxygen demand and the release of cell contents	♦ Success appears linked to uncertain and potentially uncontrollable water chemistry factors ♦ Depression of oxygen levels may result ♦ Water chemistry may be altered in other ways unsuitable for non-target organisms

3.2 NON-POINT SOURCE NUTRIENT CONTROL

3.2.1 The Nature and Control of Non-Point Source Pollution

In recent decades, with the success of the Clean Water Act and other environmental protection efforts, non-point source inputs from land use activities has become the major source of surface water pollution (MDEP, 2000). Non-point source (NPS) pollution is defined by the USEPA as pollution of surface water or groundwater supplies originating from land-use activities and/or the atmosphere and having no well-defined point of entry. Usually NPS pollution includes all sources of nutrients that do not emanate from a pipe, although the regulatory definition can include ditches, swales, or even narrow curb cuts. Because of the lack of a distinct discharge in most cases, non-point source pollution is often difficult to control. NPS pollution may include toxics, organics, heavy metals, oil and grease, turbidity, bacteria and other pathogens as well as nutrients. For the purposes of this environmental impact report, we will focus on the effects of nutrients, primarily phosphorus and secondarily nitrogen, which cause eutrophication in lakes. Suspended sediment is also a NPS pollutant and can impact lakes both by carrying nutrients to lakes and by reducing lake depth (which tends to increase the rate of eutrophication). Further discussion of sediments can be found in Section 3.7 (Dredging).

Although both phosphorus and nitrogen are essential nutrients (fertilizers) for aquatic plant growth, phosphorus is the nutrient most often associated with cultural eutrophication and the focus of most lake restoration efforts. NPS pollution is most often associated with urban runoff, agricultural operations (including crops, livestock and silviculture), forest industries (especially logging), domestic on-site wastewater disposal (septic) systems, construction activities, and a variety of other land use activities of lesser overall impact (but still potentially important in individual cases). In some cases nutrients may come from natural sources, such as high concentrations of birds such as gulls, ducks and geese. Excess nutrients have been identified as the cause of not attaining the designated use support in Massachusetts for 27 percent of lake acres that have been assessed (MDEP, 2000).

Urban runoff is often considered to be the major source of NPS pollution (MDEP, 2000), although in rural areas agricultural practices may be a greater concern for eutrophication. Urban development in watersheds has been the greatest threat (Robbins et al., 1991), however, with both construction activities and post-construction runoff as major issues. The 2000 305b report on water quality in Massachusetts indicates that only 30% of the assessed lakes in the state fully support their designated uses, and many of these are considered threatened (MDEP, 2000). Proliferation of aquatic plants and excess nutrients are cited as causative agents in over half the cases of non-support, and NPS are prominent in many of these cases. Combined sewer overflows are another significant source of nutrients, but are considered under point sources.

Commercial fertilizers are the major source of agricultural NPS nutrients. Application of P and N in the United States has increased by 3-fold and 20-fold, respectively, between the years 1945 and 1993 (Puckett, 1995). Perhaps because most farmers are more concerned with achieving adequate nitrogen levels, current fertilizer and manure application rates have led to a build up of soil phosphorus levels in the northeastern United States (Sharpley et al., 1994). Although it is not harmful to the crop to have extra phosphorus, it does result in eutrophication of surface waters. This important issue is thoroughly reviewed in Sharpley et al. (1994). In many urbanized areas of Massachusetts, the residential use of fertilizers is a larger source of nutrients than agricultural fertilizers, and the process is the same.

A review of 32 Diagnostic/Feasibility studies (Tables 1-2 and 1-3) shows an extremely wide range of values for most sources. From land use analysis, the major sources were residential. Based on measured inputs, the major transport of phosphorus to lakes was from streams, ground water and internal sources, although wide differences in inputs were observed from lake to lake. The importance of septic systems in phosphorus loading may be overstated, as attenuation in soil between the tank and the lake was seldom assessed, but septic systems remain a major source of nitrogen in lakes. Storm water runoff, assessed from only a few storm measurements, may have been underestimated in many cases. Agricultural/open field inputs accounted for a median of only about 7 percent in this survey, which may be a reflection of relatively high housing density and relatively few agricultural acres within the watersheds of the studied lakes.

Control measures are summarized under two processes, source controls and pollutant trapping. Within the pollutant trapping process, approaches applicable to urban landscapes, recreational facilities, agriculture, forestry, wastewater disposal, and non-human sources are briefly addressed. There is an extensive literature base for this subject, one that is too great to cover completely in this GEIR, and readers are encouraged to review the materials contained in a wide variety of works on the topic of NPS control. These include:

- Schueler, T. 1987. Controlling Urban Runoff: A Practical Manual for Planning and Designing Urban BMPs. Metropolitan Washington Council of Governments, Washington, DC. One of the original works calling attention to urban runoff problems and how to solve them, this reference provides a foundation upon which other references build. Many references by Schueler and colleagues can be obtained through the Center for Watershed Protection on the internet at www.cwp.org.
- Dennis, J., J. Noel, D. Miller, and C. Eliot. 1989. Phosphorus Control in Lake Watersheds: A Technical Guide to Evaluating New Development. ME DEP, Augusta, ME. While developed to provide support for meeting the development regulations in Maine, this document provides useful estimates of performance and constraints associated with detention facilities and buffers. Available from ME DEP in Augusta, ME.
- Wisconsin Department of Agriculture and Trade and Consumer Protection. 1989. Best Management Practices for Wisconsin Farms. WDATCP, Madison, WI. Prepared in cooperation with many state agencies and the University of Wisconsin Extension Service, this manual provides detailed information on why and how to manage P and N in agricultural operations. It does not appear readily available at this date, but might be obtained through the Extension Service on the internet at IPCM@wisc.edu.
- Sage, K. 1990. Implementation Strategies for Lake Water Quality Protection: A Handbook of Model Ordinances and Non-Regulatory Techniques for Controlling Phosphorus Impacts from Development. North Kennebec Regional Planning Commission, Kennebec, ME. This compendium of ordinances and operational controls provides the basis for many community efforts to manage impacts from developments. Advances since 1990 have been substantial, but this document provides a useful starting point. Available from ME DEP, Augusta, ME.
- Schueler, T., P. Kumble and M. Heraty. 1992. A Current Assessment of Urban Best Management Practices: Techniques for Reducing Non-Point Source Pollution in the Coastal Zone. Metropolitan Washington Council of Governments, Washington, DC. This review of performance of various BMPs provides much useful design information and performance data. It is available from the Center for Watershed Protection at www.cwp.org.
- Boutiette, L. and C. Duerring. 1993. Nonpoint Source Management Manual. MDEP, Boston, MA. Often called "The Mega-Manual", this guide to techniques provides a discussion of each technique and its applicability to categories of NPS pollution. Available through the MDEP, although it has been out of stock multiple times since original publication.
- USEPA. 1993. Guidance Specifying Management Measures for Sources of Nonpoint Pollution In Coastal Waters. Document 840-B-92-002, USEPA, Washington, DC. Although developed as a coastal management guide, chapters on agricultural, forestry and urban areas are highly appropriate to any watershed management effort. Can be accessed on the internet at www.epa.gov/owow/nps/MMGI/index.html.
- New Hampshire Department of Environmental Services. 1994. Best Management Practices to Control Non-Point Source Pollution: A Guide for Citizens and Town Officials. NHDES-

WSPCD – 94-2, NHDES, Concord, NH. This layperson's guide provides guidance for managing NPS pollution in a straightforward, simplified, outline form. It may be out of stock by now, but has been available from the NHDES in Concord, NH.

- Lobdell, R. 1994. A Guide to Developing and Re-Developing Shoreland Property in New Hampshire. North Country Resource Conservation and Development, Inc. Meredith, NH. This layperson's guide provides guidance for evaluating and preventing impacts from shoreland property. It may be out of stock by now, but has been available from the NHDES in Concord, NH.
- Schueler, T. 1995. Site Planning for Urban Stream Protection. Center for Watershed Protection, Ellicot City, MD. This manual describes techniques for minimizing development impacts through storm water controls. Available through the Center for Watershed Protection at www.cwp.org.
- Claytor, R. and T. Schueler. 1996. Design of Storm Water Filtering Systems. Center for Watershed Protection, Ellicot City, MD. Center for Watershed Protection, Ellicot City, MD. This manual provides detailed engineering guidance for 11 different filtering systems applicable in storm water management, along with performance and applicability information. Available through the Center for Watershed Protection at www.cwp.org.
- Kadlec, R. and R. Knight. 1996. Treatment Wetlands. CRC Press, Boca Raton, FL. This very thick textbook provides extensive background on the use of wetlands to treat storm water and wastewater. Chapters are devoted to concepts, specific pollutants of interest, system design, and system maintenance. Available through CRC Press/Lewis Publishers.
- MDEP. 1997a. Storm Water Management. Volume One: Storm water Policy Handbook, and Volume Two: Storm Water Technical Handbook. Publ. # 17871-250-1800-4/97-6.52-C.R. MDEP, Boston, MA. This two volume set lays out the premises for the MA storm water policy and provides technical support for the design of storm water BMPs to meet the policy. It is essential background and support information for storm water management efforts in MA. Available from MDEP in Boston.
- MDEP. 1997b. Massachusetts Erosion and Sediment Control Guidelines for Urban and Suburban Areas. A Guide for Planners, Designers and Municipal Officials. MDEP, Boston, MA. This guide describes conservation BMPs and is intended to be a companion to the "Mega-Manual".
- Heufelder, G. and S. Rask. 1997. A Compendium of Information on Alternative Onsite Septic System Technology in Massachusetts. A special issue of Environment Cape Cod. Barnstable County Department of Health and the Environment, Barnstable, MA. Covering a wide range of approved and experimental technologies, this compendium provides useful information for uses of such systems on Cape Cod.
- Wisconsin Department of Natural Resources. 1997a. Wisconsin's Forestry BMPs for Water Quality: A Field Manual for Loggers, Landowners and Land Managers. WDNR, Madison, WI. This manual covers BMPs for road building, timber harvesting, prescribed burning, and chemical applications to forests. Available through the WDNR, but not online.
- Wisconsin Department of Natural Resources. 1997b. Wisconsin's Forestry BMPs for Water Quality: The 1995-1997 BMP Monitoring Report. WDNR, Madison, WI. This companion report to the BMP manual provides a review of effectiveness based on actual data. Available through the WDNR, but not online.
- The Federal Interagency Stream Restoration Working Group. 1998. Stream Corridor Restoration: Principles, Processes and Practices. USEPA, Washington, DC. This publication

deals with stream corridors and how to protect and restore them. It relates to all forms of land use that impact streams and downstream lakes. Information on availability was not provided, but the USEPA website may be helpful (www.epa.gov).

- USDA-NRCS and MA Community Assistance Partnership. 1998. Innovative and Alternative On-Site Wastewater Technologies. USEPA, Boston, MA. This guide to products and services provides information on 21 disposal systems that differ from conventional tank and leaching facilities, including performance data. Availability unknown.
- USEPA. 1999. Preliminary Data Summary of Urban Storm Water Best Management Practices. EPA-821-R-99-012. USEPA, Washington, DC. This document provides background on pollutant levels in urban storm water and effectiveness of control by various BMPs. Available through www.epa.gov.
- Winer, R. 2000. National Pollutant Removal Performance Database for Storm Water Treatment Practices, 2nd Edition. Center for Watershed Protection, Ellicot City, MD. This collection and synthesis of pollutant removal data is the most up-to-date compendium of what can be expected from various storm water management techniques.
- Washington State Department of Ecology. 2000. Storm Water Management Manual for Western Washington. Publication 99-14, WA DOE, Olympia, WA. This five-volume set of documents includes volumes on minimum technical requirements, construction pollution prevention, hydrologic analysis/flow control, source controls and runoff treatment BMPs in a concise format applicable almost anywhere in the USA.
- Thornton, K. and C. Creager. 2001. Watershed Management. Chapter 6 in Managing Lakes and Reservoirs, edited by Holdren, Jones and Taggart. USEPA/NALMS, Madison, WI. This chapter covers watershed management needs and approaches from a lake management perspective. It also provides an annotated bibliography of BMP manuals for various land uses. Available through NALMS on the internet at www.NALMS.org.
- USEPA. 2002. Onsite Wastewater Treatment Systems Manual. EPA/625/R-00/008. USEPA, Washington, DC. This revised version of the 1980 manual provides details for many onsite systems, with performance standards and related data.

3.2.2 Source Controls

Source controls consist of techniques that eliminate or reduce the potential for pollutants (in this case especially nutrients) to be released from a source. The most reliable way to do this is to eliminate the source, but this may not be practical in many cases. Successful source elimination examples include the 1994 ban on the sale in Massachusetts of laundry detergents containing more than a trace amount of phosphorus, exclusion of certain land use activities from the Zone II area of contribution to water supplies, and purchase of land for open space that might otherwise be developed. Potential successes await us in the areas of lawn fertilizers, where phosphorus is almost unnecessary after a lawn has become established or where lawns are avoided (in favor of natural vegetation). Success in long urbanized areas may be elusive, however, as a consequence of both the difficulty in gaining compliance and the long-term build up (and gradual release) of phosphorus in soils. Source controls are therefore the first line of defense, but will rarely be successful as the only line of defense.

Most source control is achieved through laws, mostly local bylaws or ordinances that restrict product contents or use. Where a feasible alternative product exists, this can be a very successful approach. Note that after enough states banned phosphorus in laundry detergents in 1995, the industry ceased production of phosphate-enriched detergents altogether. Where education reveals both an environmental and economic value by source elimination, success may also be achieved. As established lawns require very little added phosphorus, homeowners should be able to save money and protect water quality while maintaining lawns. However, the cost of no-phosphorus fertilizer is not less than phosphorus-rich brands, and a cultural shift is needed to get people to put water quality ahead of their lawns or their pocketbooks.

Pet Waste Management

An additional area of residential nutrient control is pet waste management. Education is again a key element in making a cultural shift toward minimizing nutrient loading from this source. For both lawn care and pet waste management, a survey by the Center for Watershed Protection (UNEP, 1999) found that understanding of issues and options by residents was limited, while funding and staffing of programs to combat these sources were inadequate. More effective outreach was recommended, involving television, newspapers, and the internet.

Wildlife Management

Management of wildlife can be a source control effort, especially if populations are to be reduced. Most often the focus is on waterfowl, especially geese, ducks and gulls. Resident populations supported by human actions, including direct feeding and maintenance of lawns, fields or waste disposal facilities that provide food, can be a major source of nutrients on a seasonal or even year-round basis. Habitat alteration that discourages their use of a lake may also serve as pollutant trapping mechanisms, as in the case of densely vegetated buffer strips. There is a limit to this approach, however, as alteration of habitat to suppress the population of one species may help support elevated populations of other species. A focus on ecological balance is desirable in these cases. More direct techniques include scare tactics, egg addling, and hunting to reduce populations or use of an area by a local population.

Product Use Restrictions

Product use may be allowed with restrictions, as opposed to use bans. Timing of application of fertilizers is important, as is the form in which it is applied. Location of application is another form of source control that does not eliminate the source, but will minimize its escape from the initial area of use. Minimizing the use of any nutrient-laden product over any impervious surface is a valuable use restriction that does not eliminate the product or its benefits, except for deicers and other pavement treatments intended to be used on impervious surfaces. However, deicers and sand may themselves contain substantial amounts of phosphorus and can be a threat to water quality. Use of low phosphorus deicers and "clean" sand (washed to eliminate fines) is advisable. Eliminating the impervious surface may have merit in many instances, but is more of a pollutant trapping mechanism than source control.

Activity Restrictions

Source control can also involve activity restrictions. Siting of septic systems only in suitable soils and construction of dwellings outside of wetland buffers are examples. Water supplies are accorded special protection, with activities inside one-quarter mile of the supply or its tributaries restricted to minimize pollution potential. Use of gasoline engines is prohibited in some cases, and is usually forbidden within some set distance of intakes. Placement of fertilizer only outside a buffer zone around lakes and tributaries would seem to be a worthwhile addition, but a cultural shift is again needed to gain support for such an approach. Restricting activities within the watersheds of recreational lakes is a much greater stretch, and stirs up enough legal opposition to minimize the value of source control in many areas. Lake and watershed management districts, empowered entities formed in only a few places across the USA, have met with limited success in achieving source control. The Lake Tahoe Planning Authority is among the most powerful, and yet the clarity of Lake Tahoe has declined by 30% over the last four decades (Chilton, 2002).

Zoning

Zoning is perhaps the most well known form of source control on a large scale, and involves assigning activities or uses to certain areas and banning them from others. Zoning regulations are largely controlled at the town level and are thus specific to each town. Commonly, zoning regulations limit land use within certain areas of the town. The types of land use being regulated may be industrial use, housing development and on land disposal sites. By limiting such activities in areas adjacent to lakes and streams, nutrient inputs may be reduced compared to areas of unrestricted development. However, zoning has only recently been used to manage water quality, and has been more of an economic and quality of life management tool in the past. Used to benefit a town instead of a watershed, its success in protecting water quality is often limited. Where rural communities are developing and where a watershed level approach is applied, water quality-based zoning could be a major asset, but where the watershed has already become urbanized, the potential for zoning to aid water quality is limited. Enlightened zoning may move a community in the right direction over time, but is unlikely to yield major water quality benefits in the short-term and may meet with considerable public opposition.

Wastewater Diversion

Source control may include diversion of discharges considered detrimental to the receiving water. Wastewater discharges, even when treated to the maximum practical (but not possible) extent, contain so much phosphorus (>100 ppb, usually >400 ppb, and often >1000 ppb, when values <20 ppb are desirable) that diversion of such discharges has resulted in dramatic improvement of receiving water quality. This only re-locates the problem in many cases, however, and is not truly source elimination. Elimination of septic systems in favor of a wastewater collection and treatment system may have similar consequences when the discharge for the wastewater treatment facility is out of the watershed where the septic systems were located. Only if superior treatment of collected wastewater is rendered can the actual load of nutrients be decreased overall, although the location of discharge may change to the benefit of some resources and the detriment of others.

Erosion Control

A more active form of source control is erosion control. Stabilization of stream banks or sloped soils prone to erosion and a variety of hydraulic techniques for reducing peak flows and velocities may be applied. The best way to prevent sedimentation of water resources is to prevent erosion in the first place. A certain amount of sediment loss is expected even under natural conditions, but losses one to two orders of magnitude higher are common in agricultural and developed watersheds. Stabilization may include “hard” methods such as riprap, concrete, or wooden structures, or may incorporate “soft” approaches involving more porous media and plantings. Each has its place, with advantages and disadvantages relating to site conditions, future use, longevity, and cost. In general, the steeper the slope, the higher the water velocity, and the less porous the natural soils, the more likely that "hard" techniques will be needed.

Street Sweeping and Catch Basin Cleaning

The transition from source control to pollutant trapping options is represented by street sweeping and catch basin cleaning. These techniques acknowledge the build-up of nutrients on impervious surfaces and attempt to remove them before they can reach streams or lakes. To be effective, streets must be swept often. Ideally, this means before every storm, but this is neither predictable nor practical, and sweeping every few weeks is about the maximum frequency observed. Catch basin cleaning renews the trapping capacity of catch basins and prevents high flows from resuspending and moving previously trapped pollutants. This practice could also be performed regularly, but almost never exceeds a frequency of twice per year, with many catch basins going years between cleanings. An additional limitation is that a majority of pollutants are associated with the smallest particle size fractions, and these smaller particles are not effectively collected by conventional brush street sweepers or detained in conventional catch basins. There is value in street sweeping and catch basin cleaning associated with maximizing the capacity and efficiency of downstream trapping systems. However, maximized nutrient capture requires the use of vacuum sweeping equipment and advanced catch basin design with at least annual cleaning.

One problem with source control is that a potentially large portion of nutrient loads may arrive via the atmosphere from other watersheds. In a forested watershed with typical Massachusetts soils, these nutrients would be incorporated into the forest floor and only a small fraction would be carried with runoff. With increasing impervious surface area, through development of the watershed, a greater portion of these nutrients and many other airborne pollutants are washed into streams and lakes. Inability to control the sources, coupled with alteration of the watershed that limits natural trapping mechanisms, requires that we do more than attempt to minimize sources under our control to protect water quality.

3.2.3 Pollutant Trapping

3.2.3.1 Pollutant Trapping Applications

Pollutant trapping BMPs include a wide variety of techniques that may be applied to:

- Urban landscapes – buffer strips, minimizing imperviousness, advanced catch basins, swales, detention ponds and infiltration systems to capture runoff from impervious surfaces and hold that runoff until pollutants can be removed by physical, chemical and/or biological processes. Handling the potentially high volume of runoff is a key consideration. Chemical treatment of runoff for nutrient inactivation, as with phosphorus binding by aluminum, can also be applied.
- Recreational facilities – approaches much like those for urban landscapes, but with lower runoff generation and high potential to make productive use of runoff to maintain the recreational facility. Parks, ball fields and golf courses are the most common facilities addressed, and maintaining utility despite wet weather is a key factor in successful management.
- Agriculture – planting schemes, conservation tillage, manure handling systems, manure treatment systems, buffers, swales, detention ponds and infiltration systems to limit runoff

generation and keep nutrients on the farm. Demonstrating a favorable economic trade-off between runoff management and intensity of land use is a key factor in achieving success.
- Forestry – cutting schemes, road construction and drainage, buffers, swales, detention ponds and infiltration systems to minimize sediment and associated nutrient transport to streams. Facilitating access while minimizing impact on water quality is a key consideration.
- Wastewater disposal – proper siting, construction, and maintenance of an appropriate system to maximize capture of wastewater nutrients. Maximizing performance of conventional tank and leaching systems or substituting alternative advanced on-site disposal systems is essential to minimizing impact on water resources. While additives have not been documented to be especially effective, a variety of chemical and biological additives have been applied.
- Non-human sources – wildlife management, most notably for waterfowl, to minimize impacts on water quality. This may involve efforts to relocate populations or alter habitat to trap pollutants.

3.2.3.2 Technique Summary

Applicable techniques can be summarized as follows:

- Buffer strips - Buffer strips (or vegetated filter strips or grassed buffers) are areas of grass or other dense vegetation that separate a waterway from an intensive land use. These vegetated strips allow overland flow to pass through vegetation that filters out some percentage of the particulates and decreases the velocity of the storm water. Particulate settling and infiltration of water often occurs as the storm water passes through the vegetation. Buffer strips need to be at least 25 ft wide before any appreciable benefit is derived, and superior removal usually requires a width >100 ft (Dennis et al., 1989), although a well designed system can be very effective at widths <100 ft (Lee et al., 2003). Wide buffers can create land use conflicts, but creative planting and use of buffer strips can be a low cost, low impact means to minimize inputs to the aquatic environment.

- Minimization of impervious surfaces – Water quality response to runoff has been clearly linked to the portion of the watershed that is impervious. While natural surfaces such as clay soil, muck soils, and exposed rock are functionally impervious, human derived surfaces such as roads, parking lots, driveways and roofs are major sources of runoff in developing watersheds. Once imperviousness exceeds 10% of the watershed area, water quality problems are often observed, and at levels in excess of 25%, water quality impairment almost always occurs (CWP, 2003). Imperviousness can be minimized by narrowing roadways, limiting development footprints, and incorporating porous pavement wherever feasible.

- Advanced catch basins - Deep sump catch basins equipped with hooded outlets can be installed as part of a storm water conveyance system. Deep sumps provide capacity for sediment accumulation and hooded outlets prevent discharge of floatables. Catch basins are usually installed as pre-treatment for other BMPs and are not generally considered adequate storm water treatment as a sole system. Volume and outlet configuration are key features that maximize particle capture, but it is rare that more than the coarsest fraction of the sediment/nutrient load is removed by these devices.

A number of more advanced chamber designs are currently on the market. These self-contained units include an initial settling chamber for sediment removal, typically have hooded internal passages to trap oil and other floatables, and often incorporate some form of outlet pool to control exit velocity. Several rely on a vortex design to enhance sediment removal, while others rely on filtering mechanisms to augment the settling process. Such systems are most applicable as pre-treatment for other BMPs, but can trap much of the particulate nutrient load and are generally well suited as retrofits for relatively small areas in developed watersheds. Installing these devices as off-line systems may enhance nutrient removal, but their more common use as on-line pre-treatment devices can be very beneficial.

- Swales – Engineered ditches can provide detention and infiltration while transporting runoff to a planned discharge point. Use of dense vegetation and stone or wood check dams within the confines of a channel designed to handle substantial flows of runoff can slow water velocity, allow particulate nutrients to settle, and provide infiltration of a substantial fraction of the dissolved nutrient load. Less removal may occur during higher flows, but such flows do not often carry more of the total nutrient load than smaller storms in most watersheds as a consequence of the first flush phenomenon. Swales may be adequate for nutrient removal if large and long enough, but are more effective as pre-treatment devices before discharge to detention systems.

- Detention ponds – Detention ponds are basins that are designed to hold a portion of storm water runoff for at least 12-24 hours and preferably longer. Pollutant removal is accomplished mainly through settling and biological uptake, although incorporation of infiltration capacity can add substantial adsorptive capacity as well. Design features are extremely varied and depend on pollutant removal goals, regional climate, and localized site conditions. Detention facilities can be large ponds with multiple forms of aquatic habitat or small "rain gardens". Wet detention ponds are often more effective than dry detention ponds as the latter have a greater risk of sediment re-suspension and generally do not provide adequate soluble pollutant removal. Although potentially very effective, the land requirement is typically large; the area should be at least 2% of the drainage area it serves, and preferably as much as 7% of that area.

  Detention systems tend to be created wetlands, but design features that combine open water and emergent wetlands tend to provide superior nutrient removal (Kadlec and Knight, 1996). These systems maximize pollutant removal through vegetative filtration, nutrient uptake, soil binding, bacterial decomposition, and enhanced settling. Much of the effectiveness of the treatment is related to microbial action; the plants are more substrate than active pollutant removers, but removal rates are higher in the presence of plants. Detention systems are suitable for on-line or off-line treatment, but maintenance of adequate hydrology with off-line systems is necessary to support the complete wetland features that maximize effectiveness. Constructed treatment wetlands can function effectively in cold environments, mainly as a function of subsurface flow and related microbial uptake, adsorption, and filtration processes. Presence of aerobic and anaerobic conditions in sequential portions of the system is essential to reduction in nitrogen through sequential oxidation and reduction of nitrogen forms to convert organic forms to nitrogen gas.

- Infiltration systems – Infiltration systems may include trenches, basins or dry wells, and involve the passage of water through soil or an artificial medium such as a constructed berm. Particles are filtered by the soil matrix and many soluble compounds are adsorbed to soil particles. Such systems require sufficient storage capacity to permit the gradual infiltration of runoff into suitable soils or through the constructed medium (Claytor and Schueler, 1996). Pre-treatment of the runoff removes larger particles before filtration, thereby aiding in the prevention of infiltration system failure due to clogging and sediment accumulation. Phosphorus removal is maximized by infiltration, but dissolved forms of nitrogen may be only minimally affected.

  Site constraints such as shallow depth to groundwater or bedrock and poorly drained soils often limit the effective use of infiltration, so detailed knowledge of the site is essential when planning infiltration facilities. In sites with suitable conditions, off-line infiltration systems are generally preferred. One key to successful infiltration is providing adequate pre-infiltration settling time or other treatment to remove particles that could clog the interface at which infiltration occurs. Another key is having sufficient runoff detention capacity to allow delivery of runoff to the infiltration surface at a rate that maximizes performance. Both key factors can be met by combining adequate detention capacity with infiltration systems.

- Planting plans – The spatial and temporal features of planting, coupled with the actual crops chosen, can greatly affect the movement of nutrients off farm fields. Cover crops stabilize soils, and may be used as interim cover or as a supplemental crop in association with plants that grow up through the cover crop to form another layer above it. Interspersing of crops can create buffer zones such that potential nutrient losses after harvest of one crop are held by the other. The basic philosophy of the planting plan is to minimize bare soil and create buffer zones that have economic as well as ecological value.

- Conservation tillage – The pattern of plowing on a farm can be a great aid to minimizing the movement of nutrients. Contouring, terracing, and related approaches minimize the peak velocity attained by runoff and maximize infiltration of rainwater. Coupled with an effective planting plan, the quantity of runoff generated from the field can be greatly reduced; this translates into reduced nutrient loading to area waterways.

- Manure handling systems – Livestock operations have the potential to contribute nutrient loads that overshadow most other sources, and represent a distinct health hazard as well. Manures are of special concern as they are relatively high in phosphorus relative to nitrogen and attempts to meet the nitrogen requirements by application of manure may result in losses of phosphorus to the adjacent surface waters. Handling manure in a manner that limits interaction with precipitation and incorporation into runoff is essential to protecting aquatic habitats. The Natural Resource Conservation Service, NRCS, (formerly the Soil Conservation Service, SCS) suggests that manure application be kept as far away as possible from streams and lakes, and that the application of manure be avoided during the winter months when frozen soils result in large losses to the streams in runoff (Diane Leone, NRCS, pers. comm., 1995). Covered feeding areas, manure collection systems, covered storage, and proper spreading on farm fields or disposal by other means are all necessities of best management for livestock facilities.

Even with proper spreading practices, the capacity of fields to adsorb and utilize the phosphorus provided by manure is often inadequate. In Wisconsin, a practice of phosphorus build up in soils by over fertilizing has resulted in an excess of phosphorus being applied to the field in the form of manure and inorganic fertilizers (Wedepohl, 1995). This practice was designed to optimize crop production, but it results in eutrophication of sensitive waters, and this practice appears to be occurring in Massachusetts as well. For example, soils tested from silage cornfields in Massachusetts show fairly high levels of phosphorus and thus do not require large additional amounts of phosphorus fertilizers (S. Bodine, UMASS, pers. comm. 1995). Therefore, extensive and creative systems are needed to effectively manage manure from livestock operations. Recent studies suggest alum and other chemical additives may reduce phosphorus leaching from poultry litter and manure (Moore and Miller, 1994; Shreve et al., 1995). Conversion of manure to energy is a novel approach most recently advanced in Maryland.

- Chemical additions – The use of phosphorus binders has long been practiced in water and wastewater treatment, but has only recently been extended to storm water treatment, manure management, and septic systems. Aluminum compounds have been most popular, as they bind and hold phosphorus in a biologically unavailable form under the widest range of conditions. Storm water treatment systems have been developed most extensively in Florida (Harper et al., 1999), but systems are in operation in New Jersey (S. Souza, PHydro, pers. comm.,) and pilot testing has been completed in Massachusetts, albeit with limited success (ENSR, 1997a). Manure treatment with alum is being researched in several areas with high concentrations of poultry, swine and cattle (J. DeWolfe, Sear Brown, pers. comm., 2002). The addition of alum to septic tanks does not appear to have moved beyond the conceptual stage (Brandes, 1977).

- Cutting plans – Forestry operations usually involve cutting trees, and loss of this vegetation destabilizes the forest ecosystem and often results in increased nutrient losses. A proper cutting plan can minimize those losses, and incorporates thinning instead of clearcutting, cutting patterns that maintain buffer zones, and use of waste vegetation to stabilize cut areas to the extent possible.

- Road construction and drainage – Although road management is important in all areas, proper construction and maintenance of logging roads is critical to minimizing nutrient inputs from forestry operations. Access is essential to the industry, but the temporary roads often represent a major threat to nearby streams. Road routing, slope, surface treatment, and drainage characteristics are important features. The basic objective is to prevent erosion and to direct runoff into stable areas for detention or infiltration. Stream crossings need to be stabilized to the maximum practical extent.

- Conventional septic systems – Most on-site domestic sewage treatment consists of either the older cesspool (single chamber, open bottom pit type, no longer in construction) or the newer septic tank with leaching field or chamber. Most septic systems consist of a subsurface chambered tank where scum and settleable solids are removed from the liquid by gravity separation, and a subsurface drain system where the clarified liquid effluent percolates into the soil. Regular inspection of the system is recommended, with pumping as experience dictates or according to calculations based on the number of people served and the size of the tank.

For conventional septic systems, the management techniques are detailed in Title 5 of the State Environmental Code 310 CMR 15.00 et. sec. (see State Sanitary Code in Appendix II). These regulations were revised in 1995, but allow older cesspools (pit type sewage tanks with open bottoms and sludge retention in the pit) to remain unless they fail inspection criteria outlined in CMR 15.00. Any new septic systems must comply with design and construction standards given in 310 CMR 15.00, which specifies the leach field must have a minimum setback of 50 feet from surface waters. To protect resources, additional restrictions on septic systems may be imposed by local ordinance.

Phosphorus is removed to a moderate degree in both the septic tank and the leachfield, owing to chemical reactions that tend to convert phosphorus into particulate forms. Even beyond the leaching field or chamber, soils adsorb phosphorus at high rates. With low adsorption rates on the order of 1 microgram per gram of soil, even sand will capture much of the phosphorus load from a septic system as long as the soil is aerated and past loading has not exhausted the adsorptive capacity. Where the system is in fractured rock or compacted soil with fissures, such removal may not be realized. Likewise, where system failure results in a breakout of septic effluent at the ground surface, removal of phosphorus will be severely reduced. Yet overall, the data from the more detailed D/F studies (those involving direct measurement of inseepage quality) indicate very limited impacts from phosphorus loading from septic systems on most lakes. Septic systems should be managed for long-term successful operation, but it should not be assumed that they are major sources of phosphorus without supporting data.

However, even a properly sited, well-maintained, conventional septic system will release a substantial amount of nitrogen into the ground. Much of the nitrogen in a septic system is in a soluble form or becomes soluble. Physical and chemical soil processes do little to reduce discharge concentrations, which may exceed 50 mg/L. Site limitations and the inability of conventional septic systems to capture more than about 10% of the nitrogen load has fostered a variety of alternative systems (Heufelder and Rask, 1997; USEPA, 2002).

- Advanced on-site wastewater disposal systems – In cases where a septic system fails and/or the site can not accommodate a conventional system due to size or performance needs, there are many approved alternate technologies for septic systems in Massachusetts. These include the recirculating sand filter, composting toilets, Bioclere system, Eljen in-drain system, Environchamber, Ruck system, and the Saneco intermittent sand filter, to name just a few. More recently tested systems include the Smith and Loveless system, the Amphidrome process and the Krofta compact clarifier (MDEP, 1995). A variety of concepts are put to work in these systems, but most take advantage of multi-stage processes to enhance the treatment methods at work in conventional systems or add new treatment approaches. Some systems are designed to enhance infiltration in low permeability sites, but most focus on achieving better overall effluent quality.

Many of these are designed to remove nitrogen, but apparently none have a demonstrated ability to remove significantly more phosphorus than conventional systems. This is not surprising, given the generally acceptable performance of conventional systems with regard to phosphorus. Research into enhancing phosphorus removal in septic systems is ongoing.

- Habitat alteration for wildlife control – As waterfowl like geese and ducks are often the primary target of such management, and these animals prefer open shorelines with easy access from water to land and vice versa, control is often a function of dense buffer establishment. This is entirely consistent with buffer use for runoff control. Additional controls often involve choosing plantings that decrease habitat value for species considered undesirable. Elimination of habitat is almost never the goal, as this leads to conditions that facilitate other undesirable inputs (e.g., clearcutting with impervious surfaces or fertilized lawns). Ecological balance should be sought, although this is sometimes an elusive concept and is rarely a stable condition.

3.2.4 Effectiveness

3.2.4.1 Short-Term

In general, NPS nutrient control techniques are not expected to be effective in controlling lake eutrophication in the short-term. The soils and groundwater may have high levels of nutrients that continue to runoff or leach into the lake even if source controls and pollutant trapping are fully and properly implemented. Zoning is basically a preventive measure to reduce NPS pollution before it is created. NPS control methods are not considered as effective short-term treatments for eutrophic waters.

3.2.4.2 Long-Term

NPS controls are intended to provide long-term benefits with proper implementation and maintenance. Strong and widespread evidence for long-term effects is not common, but then most NPS control efforts are no more than three decades old and are still gaining momentum. The case for long-term benefits is supported by data for drainage areas where NPS pollution is a dominant influence and has been addressed to a substantial degree, but the record is by no means complete enough to make strong generalizations about effectiveness; the range of results is quite large. Based on the many publications reviewed, Table 3-2 presents expected removal rates for N and P for various storm water management devices. Variability can be high, but the values do provide a general impression of the removal rates achievable. Techniques not listed in Table 3-2 are more difficult to evaluate in terms of effectiveness.

<u>Documentation Issues</u>

Although NPS pollution is thought to be the major source of nutrients to surface waters in Massachusetts, evaluation of management results has been a shortcoming in many lake and watershed management programs. The title of a 2001 report by a committee to congress, "Better Data and Evaluation of Urban Runoff Programs Needed to Assess Effectiveness" (GAO, 2001), also acts as a concise summary. The problem stems partly from scale of application, as NPS pollution is widespread by nature in most watersheds and NPS controls have not been exercised on a watershed-wide basis. On a smaller scale there are plenty of examples of BMPs reducing nutrient loading, but typically other sources of nutrients within the watershed of the lake still contribute to eutrophication in the lake(s) downstream.

The recently instituted demonstration projects in lake management, sponsored by the Executive Office of Environmental Affairs, represent a more comprehensive effort to apply NPS controls on a whole watershed basis and to document the results. The Long Pond (Littleton, MA) project involves a variety of Low Impact Development techniques being applied on a larger scale than in most watersheds (S. Roy, GeoSyntec, pers. comm., 2002).

In many cases it is difficult to discern changes in lake water quality resulting from NPS controls in only a part of the watershed. This is a function of both the level of monitoring needed and the detection limits often achieved for phosphorus. Livestock and related agricultural BMPs applied to the Pontoosuc Lake watershed appear to have reduced inputs to area streams, but any changes in the lake itself could not be detected at the relatively low levels of phosphorus observed (<40 ppb in most samples, pre- and post-application) with only a few storms sampled (ENSR, 2000a).

Table 3-2 Range and median () for expected removal (%) for nutrients by selected management methods, Compiled from literature sources for actual projects and best professional judgment upon data review by K. Wagner.

	Total P	**Soluble P**	**Total N**	**Soluble N**
Street sweeping	5-20	<5	5-20	<5
Catch basin cleaning	<10	<1	<10	<1
Buffer strips	20-90 (30)	10-80 (20)	20-80 (30)	0-62 (5)
Porous Pavement	28-85 (52)	0-25 (10)	40-95 (62)	-10-5 (0)
Conventional catch basins (Some sump capacity)	0-10 (2)	0-1 (0)	0-10 (2)	0-1 (0)
Modified catch basins (deep sumps and hoods)	0-20 (5)	0-1 (0)	0-20 (5)	0-1 (0)
Advanced catch basins (sediment/floatables traps)	0-19 (10)	0-21 (0)	0-20 (10)	0-6 (0)
Vegetated swale	0-63 (30)	5-71 (35)	0-40 (25)	-25-31 (0)
Infiltration trench/chamber	40-100 (65)	25-100 (55)	35-80 (51)	0-82 (15)
Infiltration basin	38-85 (62)	35-90 (60)	22-73 (52)	-20-45 (13)
Sand filtration system	21-95 (58)	-17-40 (22)	19-55 (35)	-87-0 (-50)
Organic filtration system	23-99	5-76	29-65	-20-10

	(65)	(40)	(46)	(0)
Dry detention basin	13-56	-20-5	10-60	0-52
	(27)	(-5)	(31)	(10)
Wet detention basin	12-91	8-90	6-85	0-97
	(49)	(63)	(34)	(43)
Constructed wetland	0-97	0-65	23-60	1-95
	(55)	(30)	(39)	(49)
Pond/Wetland	24-92	1-80	0-83	9-70
Combination	(63)	(42)	(38)	(34)
Chemical treatment	33-95	45-95	19-85	0-22
	(70)	(80)	(50)	(10)

Likewise, the reduction in phosphorus loading to Lake Lorraine after multiple storm drains were tied into leaching chambers was apparent from individual storm discharge data, but the change in the lake during the first year of infiltration was not statistically significant for this relatively clean lake (ENSR, 1997b). Claytor and Brown (1996) provide a framework for evaluating the success of NPS programs aimed at storm water, including physical, chemical and biological measures. For lakes, water quality indicators and biological indicators are suggested as the most useful evaluation factors, applied close to the management action as well as in the target lake.

Urban Runoff BMPs

Where the portion of the watershed subject to NPS controls has been monitored, or the NPS pollution addressed is the dominant influence on the lake, evidence of success has been gathered. The Emerald Square Mall storm water management project in North Attleboro, MA, included over 10 years of discharge monitoring that revealed water quality suitable for discharge into a drinking water supply (D. Lowry, ENSR, pers. comm., 2000; unpublished DMR data on file with USEPA). There is a great deal of literature documenting reduction of nutrient export in streams associated with BMPs (Schueler, 1997; WDNR, 1997b, USEPA, 1999; Winer, 2000). The actual amount of nutrient reduction will depend on the type of treatment and the percent of the watershed under management. In general, BMPs are thought to be effective at preventing further deterioration and enhancing the effectiveness of other nutrient control techniques.

Effectiveness of techniques aimed at urban storm water is highly dependent upon the extent and details of application. Urban BMPs such as porous pavement can reduce TP by 40 to 80% (Schueler, 1987). However, Schueler et al. (1992) caution that some BMPs such as porous pavement have poor longevity unless well designed and maintained. Generally, street sweeping does not produce significant reductions in pollutants such as phosphorus that are associated with fine particles (Robbins et al., 1991), but is a valuable pre-treatment step before infiltration, which can potentially decrease phosphorus levels by over 90%. Riparian forests have been shown to reduce groundwater nitrate concentrations by 36 to 80 percent (Simmons et al., 1992). Peterjohn and Correll (1984) found riparian forests to be effective at trapping nutrient runoff and leaching from agricultural fields. However, buffer width of at least 15 meters was determined to be necessary under most conditions (Castelle et al., 1994). Welsh (1991) provides a useful overview of buffer effectiveness and controlling factors.

In terms of overall effectiveness, a study by the Center for Watershed Protection (Caraco et al., 1998) indicated 46 to 60% reduction in P and 42 to 46% reduction in N when storm water BMPs were applied (termed “innovative site design”) versus a conventional development approach. The study included comparisons of medium density residential, rural subdivision, shopping center, and office park developments, and also found that the cost of development was lower by 5 to 20% with the innovative site design techniques. Much of the benefit was derived by the 18 to 35% reduction in impervious cover.

Agricultural Runoff BMPs

Agricultural BMPs generally do not show dramatic reductions in loading except in small subwatersheds where phosphorus loading has been shown to be reduced by 26-44 percent (Meals, 1993). Stafford Pond, a drinking water supply and recreational resource in Tiverton, RI, was impacted by a dairy farm in one small sub-watershed. That dairy farm was subjected to operational and structural controls, and conditions in Stafford Pond improved dramatically over a single year (ENSR, 1997c; K. Wagner, ENSR, pers. obs., 1996-2000; RI Watershed Watch Program, unpublished data, 1990-2000). Preliminary results from agricultural BMPs in the Lake Murray watershed of South Carolina suggest significant reductions of nutrients entering the lake (USEPA, 1996).

Participation can be a major factor in agricultural BMP success. Another study of the effectiveness of BMPs found that low participation by farmers in the project area prevented the BMPs from achieving the desired phosphorus reduction goals (Johengen et al., 1989). A five year study in Lake Hermon, South Dakota found that voluntary BMPs (sediment control structures) reduced sediment and nutrient loads. However, no reduction was seen in the lake nutrient levels even though 87 percent of the land area had been treated with BMPs (Payne and Bjork, 1984). At some sites Payne and Bjork (1984) found that sediment control structures did reduce concentrations of total phosphorus and inorganic nitrogen, but the cumulative effect was not measurable within the constraints of the program. In theory, nutrient management programs can be effective at reducing nutrient loadings from agricultural fields to surface waters while saving fertilizer costs. A review of agricultural pollution reduction programs indicates reductions of 22.8 to 84.2 pounds per acre for phosphorus and 11 to 44.7 pounds of nitrogen per acre (USEPA, 1993).

<u>On-Site Wastewater Disposal Systems</u>
Septic systems are most effective at reducing nutrients when they are properly sited, designed and maintained, but even then the control of nitrogen is limited. The potential for groundwater and surface water contamination increases as the density of septic systems increases (Scalf et al., 1977). A study by the USGS on Cape Cod (Persky, 1986) found a strong correlation between housing density (and by extension, septic system density) and nitrate levels in the groundwater, but also found septage disposal sites and fertilization practices to be influential. This study found no relation of sodium, ammonium, or pH to housing density. Septic systems may fail if underdesigned or if they are constructed in areas where soils are not sufficiently permeable and where the water table is too high (Robbins et al., 1991). If not pumped regularly (typically about every 2 to 4 years) solids may accumulate in the tank and eventually clog the leach field, causing failure. In such cases the effluents may appear at the ground surface and reach surface waters with runoff.

Nitrogen is higher in concentration than phosphorus in septic tanks, averaging about 40 mg/L for total nitrogen and reaching levels as high as 70 mg/L. Septic tanks are generally ineffective at removing nitrogen, but generally cause conversion of organic nitrogen to ammonium ions (Cantor and Knox, 1985). The ammonium is then converted to nitrate in the soil and may affect nearby lakes. In zones designated as nitrogen sensitive areas (see 310 CMR 15.215), regulations limit the size of new systems. For details, exceptions and information on enhanced nitrogen removal systems see 310 CMR 15.214 et seq. Removal of about half the nitrogen is possible on average with advanced systems, with removal rates ranging from 22 to 80% (Heufelder and Rask, 1997)

In Massachusetts, phosphorus content of laundry detergents was restricted as of 1994 (105 CMR 680). Estimates of potential reduction in phosphorus loading to lakes from a reduction of phosphorus in septic system effluent range from 0 to 25%, with an average of 8.5% (IEP and Walker, 1991). However, it was noted that estimation of attenuation of phosphorus between the leachfield and the lake may not have been adequate, suggesting that these estimates are high. Follow-up analysis of more recent data for lakes used in the IEP/Walker study indicates that no change in phosphorus loading from septic systems is detectable as a result of the phosphate detergent ban, mainly as a consequence of monitoring and methodological constraints (ENSR, 2003).

Untreated domestic wastewater may contain between 4 and 15 mg/L of total phosphorus (Metcalf and Eddy, 1979). Much of this is in particulate form and will be trapped in the tank. It would not be unusual to find 1 to 3 mg/L in the wastewater entering the leaching system, however. The concentration of phosphorus entering the leachfield is reduced by about 50% by the adsorption system in the immediate vicinity of the leachfield (Cantor and Knox, 1985), and will decline further in accordance with soil properties as the effluent moves through the soil. As phosphate is adsorbed onto iron and aluminum oxides in acid and neutral soils and calcium tends to bind with phosphates in alkaline soils, transport to surface waters may not be a major concern in most cases.

However, when septic systems are located near a water body or in sandy or gravelly soils with poor cation exchange capacity, there may be transport to nearby surface waters. The variability in potential impacts is too great to depend upon generalizations from the literature, and site specific studies are needed. A review of septic system impacts at the watershed level (Swann, 2001) indicates a lack of scientific support for claims of major impacts or lack of impact. Septic systems are believed to be a major source of coastal pollution and groundwater contamination, but this is a function of nitrogen discharge, and there is little evidence to suggest that septic systems contribute appreciably to phosphorus loading at the watershed level.

<u>Preventive Zoning</u>
The effectiveness of zoning is difficult to assess because zoning laws differ in each town and zoning is by nature a preventative measure rather than a treatment. In this respect, zoning effectiveness at reducing lake eutrophication is similar to BMPs; the effectiveness will depend on the methods and extent of application. One method to assess the effectiveness of zoning is by modeling the export of nutrients under different land-use scenarios as suggested by Harper et al. (1992). Reviews of nutrient export studies indicate that urban areas export the most phosphorus per unit area (0.1 $g/m^2/yr$), followed by agriculture at 0.05 $g/m^2/yr$ and forests at 0.005 to 0.01 $g/m^2/yr$, while atmospheric deposition directly to the lake is estimated at 0.025 $g/m^2/yr$ (Rast and Lee, 1983). The same study reported that nitrogen export rates show smaller differences, with both urban and rural areas exporting 0.5 $g/m^2/yr$, and forests exporting 0.3 $g/m^2/yr$. These rates suggest that the best zoning strategy to reduce eutrophication is to limit urban development and promote forest lands. Simple calculations suggest how much phosphorus and nitrogen can be expected to be reduced under any proposed zoning regulations.

3.2.5 Impacts to Non-Target Organisms

Few adverse impacts of using best management practices to reduce non-point source pollution are expected. Care must be taken at the point of application (e.g., where a detention basin, swale, or buffer strip is constructed) not to disturb sensitive habitat, especially for protected species.

Most techniques have a tendency to create or protect habitat, however, and are perceived as beneficial to the overall ecosystem. Constructed detention areas may provide valuable wetland habitat, although the primary function of these systems should not be forgotten. Buffer strips provide habitat for a variety of species in addition to water quality benefits. Reduction in impervious surface preserves habitat. Methods for non-point source pollution abatement are generally intended to enhance conditions for a majority of species.

However, reduced nutrient loading means lower overall fertility in the receiving lakes, which usually means lowered fish production. This may produce a ripple effect throughout the lake ecosystem, with quantitative decreases in species that eat fish, such as certain waterfowl and mammals. Qualitative aspects of the ecosystem are expected to improve, and may offset any quantitative losses, but lower productivity is not consistent with all management goals.

In some cases there may be temporary and limited adverse effects such as increased erosion during construction of structural controls for erosion (e.g. terraces) or in construction of manure holding tanks, but these adverse impacts are small in comparison to the expected benefits. In the long-term, most non-target organisms should benefit from most of these methods. As with any management program, however, there may be some trade-off between habitats and species, and the goals of the project should be stated clearly and be consistent with regulatory constraints.

3.2.6 Impacts to Water Quality

There may be some short-term increase in suspended sediments during construction of structures, but this is expected to be small and easily controlled in most cases. Over the long-term, water quality should improve, although it may take more than a year to discern improvements.

3.2.7 Applicability to Saltwater Ponds

Although no literature is available on the use of NPS nutrient controls specifically for saltwater ponds there is no reason that they could not be used successfully. The USEPA (1999) manual for management measures for NPS in coastal waters contains considerable applicable guidance. Saltwater ponds may be limited by nitrogen rather than phosphorus and nutrient testing should be conducted prior to beginning NPS nutrient reduction. If the emphasis is to be placed on nitrogen, the relative value of many techniques may be altered (Table 3-2). In many cases, septic system management will become a prime concern, as on-site wastewater disposal is a major nitrogen source. Improving septic systems may also improve the possibility for shellfishing in saltwater ponds due to the reduction in fecal coliform bacteria levels.

3.2.8 Implementation Guidance

3.2.8.1 Key Data Requirements

Data requirements for this type of nutrient control include an accurate nutrient budget including both a measured mass balance and a land-use source analysis. Nutrient budgets should include analysis of all inputs, including internal sources (recycle within the lake, Section 1). Nutrient control should target enough of the load to attain the desired reduction in loading to the lake, with estimates of effectiveness made for lake recovery in terms of total phosphorus levels and

Secchi disk transparency. Models of watershed loading and lake response are helpful in this regard, but only mimic reality; the use of several modeling approaches is recommended.

For most structural techniques in NPS control, knowledge of the expected water load is essential to proper sizing and other treatment considerations. Systems must have adequate capacity to handle inflows up to the point at which lowered treatment efficiency is not considered a problem for achieving nutrient loading reduction goals and the system itself will not be damaged. Undersizing NPS controls is the primary cause of failure to achieve treatment objectives. Flow considerations include total volume, distribution of volume over time, peak flows, and the distribution of nutrient loads in the flow over time. Storm water is notoriously variable in quantity and quality, but effective treatment must account for that variability.

Because of the potential for long-term benefits and minimal adverse impacts, non-point source nutrient control should be encouraged in the watersheds of all lakes. However, additional techniques may be necessary to achieve desired conditions.

3.2.8.2 Factors That Favor This Approach

The following considerations are indicative of appropriate application of NPS controls for reductions in nutrient concentrations in lakes:

1. A substantial portion of the P and/or N load is associated with NPS pollution.
2. Studies have demonstrated the impact of identifiable sources (e.g., piped runoff, septic systems) on the lake.
3. Water associated with NPS inputs is important to lake hydrology.
4. Sizing and pollutant removal functions have been properly calculated.
5. Jurisdiction can be claimed over areas of NPS contribution.
6. Land is available for placement of BMPs.
7. Detention capacity is available to hold a substantial portion of the targeted runoff.
8. Detention and/or infiltration will not cause local flooding problems, wet basements, or structural damage.
9. Infiltration will not cause groundwater quality deterioration.
10. Zoning or other restrictions on uses of land are properly justified and consistent with applicable state and local laws.

3.2.8.3 Performance Guidelines

<u>Planning and Implementation</u>

Perhaps the most significant problem involved with BMPs is convincing farmers and landowners that it is in their best interest to reduce non-point source pollution. Usually an agency such as the Natural Resources Conservation Service works with farmers and landowners on a one to one basis, but this takes time and it is difficult to get all of the landowners in the entire watershed to cooperate. Major educational efforts appear essential to success, but are rarely supported to the extent necessary. Implementation of NPS control on a watershed level is therefore typically a slow process of many small steps. Monitoring on the proper spatial and temporal scale is essential to demonstrating success, and success builds upon success. Long-term vision, public relations and funding are as important as science in accomplishing NPS control goals.

An effective lake association can help in getting property owners to reduce NPS pollution. The area adjacent to the lake is the most critical for phosphorus loading in most cases, as there is less opportunity for pollutant trapping. Simple measures such as not using phosphorus fertilizers on lawns and using phosphorus free cleaning agents may reduce the local sources of phosphorus to the lake. More involved actions such as buffer strip creation and maintenance, installation of detention or infiltration systems, and minimization of impervious surface area are likely to provide even greater benefits, but may be more difficult to achieve. Funding incentives are often needed.

Upgrading septic systems can be expensive. Regular maintenance and pumping add additional costs, and surveys of watershed residents during many of the D/F studies of the 1980s suggested that most systems are not properly maintained. The new Title V creates disincentives for allowing systems to fail, and educational programs have raised awareness in specific watersheds, but more effort is needed. Nitrogen loading can be modeled fairly reliably as a consequence of many past studies. However, as the impact of septic systems on phosphorus loading is highly variable, lake-specific studies may be needed to determine the value of septic system management or alternative wastewater disposal arrangements. The economic impact of a failed septic system is severe enough to warrant at least regular inspection and pumping (1-4 years depending on size and use features), and cooperation in this regard should be largely a matter of public education.

Zoning regulations can be seen as excessive governmental control over private lands, and thus are often resisted by landowners. Where development has already occurred, this may not be a very fruitful approach, but when a rural area is expected to be developed, a master plan can preserve the very characteristics that make the community so desirable for development.

<u>Monitoring and Maintenance</u>

For many of the BMPs to remain effective, they must be maintained. However, design of many BMPs attempts to minimize the frequency of maintenance needed. Most detention facilities need not be cleaned out more than once every 5-10 years. Most catch basin systems should be inspected at least annually, with annual cleaning likely to be necessary. Infiltration systems will lose capacity and eventually clog, with maintenance frequency dependent upon loading characteristics. For each NPS project, a monitoring and maintenance plan should be developed at the start.

Monitoring of the nutrient concentrations in water entering and leaving areas under management by BMPs should be conducted both before and after implementation to estimate the effectiveness of the BMPs. Monitoring should also be conducted in the lake to measure effectiveness on an ongoing basis, but immediate improvement with small scale BMPs in any but the smallest watersheds should not be expected.

<u>Mitigation</u>

No mitigative measures are typically required except erosion control during construction of structures, but there are exceptions. For example, the application of phosphorus inactivators to storm water may require pH control, and deposition of residuals formed by precipitation has been raised as a concern. Again, monitoring on appropriate spatial and temporal scales is necessary to assess any need for mitigation.

3.2.9 Regulations

3.2.9.1 Applicable Statutes

Runoff BMPs

In general there are no regulations that limit the use of BMPs, unless work is being conducted in wetland resources or alters them by discharge. There are exemptions in the Wetlands Protection Act for some agricultural activities (see WPA, Appendix II). In cases where wetlands are being threatened by erosion, nutrient runoff or other poor land management techniques, the local Conservation Commission may require BMPs under the Wetland Protection Act. The Massachusetts Storm Water Policy provides guidance on types of BMPs and targets for runoff control in association with new development or re-development. Provisions of this policy generally encourage infiltration, but may restrict it if certain pollutants are involved and will increase costs in most cases (simple infiltration without any pre-treatment is discouraged).

BMPs are required during forest harvesting to reduce NPS pollution as stated in MGL C. 132 s.40-46 the Massachusetts Forest Cutting Practices Act. An approved Forest Cutting Plan is required and a Notice of Intent must be sent to the Conservation Commission with a copy to the Department of Environmental Protection Regional Office. If the proposed project occurs within an Estimated Habitat of Rare Wildlife in the most recent version of the Natural Heritage Atlas, a copy of the Notice of Intent must be submitted to the Natural Heritage and Endangered Species Program (NHESP) within the MDFG for review (Appendix II). If the proposed project occurs within a Priority Habitat of Rare Species in the most recent version of the Natural Heritage Atlas, the project proponent must submit project plans to the NHESP for an impact determination. Notification must be mailed to abutters within 200 feet of the harvest area (Kittredge and Parker, 1995).

Septic Systems

Under Title 5 approved septic systems are required for sewage disposal in areas where sewer connections are not available. For details see the State Sanitary Code (Appendix II).

Zoning

Zoning regulations differ in each town. Generally, most towns have minimum limitations on lot sizes. The state has passed further restrictions on development within 200 feet of rivers and streams (Rivers Protection Bill, Appendix II).

3.2.9.2 Impacts Specific to Wetlands Protection Act

The following overall impact classification is offered as a generalization of impacts, with clarifying notes and caveats as warranted:

1. Protection of public and private water supply – Benefit (surface water quality enhanced), but possible detriment to groundwater from infiltration.
2. Protection of groundwater supply - Benefit to neutral (reduced runoff may mean greater infiltration), but possible detriment to groundwater from infiltration (groundwater quality issues).

3. Flood control - Benefit (reduced runoff volume or detention capacity increase).
4. Storm damage prevention - Benefit (based on flood control and erosion avoidance).
5. Prevention of pollution - Benefit (water quality enhancement).
6. Protection of land containing shellfish - Neutral, but possible benefit through water quality enhancement.
7. Protection of fisheries - Benefit (water quality enhancement), but possible detriment through reduced fertility.
8. Protection of wildlife habitat – Benefit (water quality enhancement and habitat creation), but possible detriment through reduced fertility.

3.2.10 Costs

The actual costs of the BMPs vary depending on the method, and it would be best to express costs on the basis of $/lb of P removed or $/acre of watershed. This has rarely been done, however, making cost comparisons difficult. While cost is always a factor in resource management decisions, in NPS control efforts it may be better to decide first on the most appropriate approach and then work out the costs.

Perhaps the most effective, simple and inexpensive BMP is a simple soil test for nitrogen and phosphorus. The soil test requires only a cup of soil (preferably collected by combining many small samples from various areas of the field), and usually costs about $10 per sample. The Soil and Plant Testing Laboratory at the University of Massachusetts will test the soil and make fertilizer recommendations for the lawn or farm. Other simple BMPs that carry minimal cost include use of non-phosphate fertilizers, leaving a buffer zone along a stream (where it already exists), minimizing impervious surface (this may also save money in construction), conservation tillage, following a forest cutting plan, and inspection of septic tanks. Pumping of septic tanks typically costs $150 to $250, a small cost compared to the expense of system replacement after failure (typically >$4000 and possibly as high as $20,000).

Zoning regulations do not cost money per se, but they may have economic impacts on land values and property taxes, depending on how they are implemented. Likewise, costs associated with passage of local ordinances are largely internalized within the community, but the economic impact is highly variable and may be substantial. Educational efforts have frequently been conducted in watersheds for $2000 to $20,000, but these have been cited as underfunded relative to the need.

The cost of structural controls is somewhat site specific, and any estimate of "typical" costs will have a high confidence interval around it. Detention basins tend to cost on the order of $20 per cubic yard of detention, with the volume of needed detention dependent upon watershed size and expected storm water flows. For a detention basin serving a 10-acre drainage area with type C (moderately low permeability) soils and a design storm of 2 inches, the target capacity would be around 1750 cy at a cost of about $35,000. A lower cost may be possible if the topography minimizes excavation needs, and a higher cost might be incurred if there are issues with ledge or discharge to sensitive receiving waters. Land costs are also extra, and may be substantial.

Infiltration structure costs depend largely on needed capacity, which is a function of both the rate of incoming runoff and the infiltration rate of the soil or bermed medium. Simple manhole or catch basin replacements, which involve installing a perforated chamber to allow leaching before overflow, can be put in place for about $6,000 to $10,000 each. A drainage system for a typical residential street might be served by 1-3 of these leaching structures. More sophisticated offline systems, whereby runoff up to some design capacity (preferably the first 0.5 inches or more of runoff from the drainage area) is diverted to a chamber or set of pipes for infiltration, can be considerably more expensive. Such an arrangement for the same "typical" residential street might cost more than $50,000, and might require an additional detention area if the soils are not porous enough to keep up with runoff generation.

Combined systems (e.g., pond/wetland, detention/infiltration) may cost as much as the individual total for each system, but some economy of scale and design is often achieved. Overall costs for projects provide some feel for the range of likely costs. The simple leaching chambers installed in drainage systems discharging into Lake Lorraine in Springfield cost about $10,000 each in 1996 and served about 5 acres each. The rain gardens planned in conjunction with the demonstration project in the watershed of Long Pond, Littleton, cost about $1,000 each and handle up to a half acre of drainage area. The advanced grease and grit trap and wet pond/wetland combination at Hills Pond in Arlington cost about $45,000 and served 10 acres. The detention ponds and constructed treatment wetlands serving about 250 acres at the Emerald Square Mall cost nearly $2 million. Capital costs therefore ranged from $2,000 to $8,000 per acre served for these projects. Maintenance costs have been limited in each case, but periodic removal of accumulated sediment is needed. Annualized maintenance costs of perhaps $50 to $100 per acre served might be expected.

On a large scale within a watershed, it might be possible to push the capital cost down to around $1,000 per acre served, but lower costs than this should not be assumed. For large watersheds like that of Lake Massasoit in Springfield, at 34 square miles of urban/suburban land use, the costs become very high and progress is slow. NPS control is best practiced near the source; costs will be minimized and localized. Attempting to handle NPS pollution at just a few locations within a larger watershed may require an area as big as or bigger than the lake being protected. In the case of Lake Massasoit, the 200-acre lake is less than half of the minimum size it needs to be to provide adequate detention of runoff from the 22,000-acre watershed. Competition for the lowest cost watershed management solution to eutrophication problems in Lake Massasoit by the Springfield College watershed management class has been won in recent years at levels ranging from $50-100 million. In contrast, an alum dosing station on each of the two major inlets could be constructed for about $2 million and operated for about $1 million per year.

3.2.11 Future Research Needs

NPS management programs need to conduct scientifically credible monitoring of surface waters both before and after implementation in order to assess effectiveness in a manner that can guide future planning. Current estimates are useful (Table 3-2), but variability in results is rather high. Additional studies should be conducted to assess control of phosphorus inputs from manures and commercial fertilizers, as these are potentially very large sources. Actual inputs from septic systems may require assessment on a lake-specific basis to determine management needs and efficacy. Additional research is needed to determine the effectiveness of alternate sewage disposal systems in reducing nitrogen and phosphorus inputs to lakes. More research into alum dosing of storm water in cold climates is needed to determine the potential for this technique in this region. Cost estimation on a consistent scale is needed to facilitate realistic economic planning and cost-benefit analyses.

3.2.12 Summary

NPS controls are recommended as preventive measures for all lakes to reduce eutrophication rates. In cases where NPS nutrients are identified as the major source of nutrients to the lake, the discussed measures are necessary, but they may have to be combined with in-lake treatments such as dredging or alum treatment to effect a recovery to desirable conditions. There are no serious ecological disadvantages to the application of storm water controls, the sensible implementation of zoning laws, or septic system management/upgrade. The advantages of most BMPs are that they are environmentally sound conservation practices often in the best interest of the land owners. The most significant drawback is that these measures are difficult to implement on a watershed wide scale, mainly as a function of cost. Partial implementation of BMPs may achieve local reductions in nutrient export, but if other sources are significant, there may be no observable improvement in the lake. NPS controls are intended to prevent problems, not reverse them, although some measure of ecosystem recovery can be expected with successful NPS control.

3.3 POINT SOURCE NUTRIENT CONTROL

Point source pollution is defined as originating from a pipe or other distinct conveyance under federal regulations. Originally intended to deal with wastewater treatment discharges from industrial or municipal operations, the definition of a point source was extended in 1990 to include storm water discharges where the delivery was an observable pipe, ditch, swale, curb cut, or other delivery device that could be construed as meeting the federal definition. Certain activities, such as concentrated animal feedlot operations (CAFOs), have also been classified as point sources in this manner. This piece of legal maneuvering created the federal storm water program under the National Pollutant Discharge Elimination System, or NPDES. Many states have been authorized to administer this program, but Massachusetts is still governed by the federal program and does not issue NPDES permits by itself. The MDEP is involved in NPDES issues, however, jointly issuing NPDES permits with the USEPA and providing considerable guidance on meeting NPDES requirements.

For the purposes of this GEIR, storm water has been addressed under NPS pollution, so the focus of this section will be actual wastewater treatment facility (WWTF) discharges. Storm water will be addressed where the NPS discussion did not cover salient point source issues.

3.3.1 The Nature and Control of Point Source Pollution

Although industry and other activities may have point source discharges of pollutants, most of the nutrient sources are from municipal WWTFs and the discussion here will focus on this type of point source. Advanced wastewater treatment as a lake management technique has been a difficult and expensive endeavor which is currently enjoying renewed vigor as a consequence of USEPA scrutiny of NPDES permits that have come up for renewal. In general, however, improved treatment has not been overly successful to date in making a marked difference in lake condition. This is a consequence of treatment limits and the high influent P levels in WWTFs, relative to the rather low levels necessary to constrain productivity in most Massachusetts lakes. As a result, control of point source nutrient loading has in some cases involved diverting the discharge away from the lake (see Section 3.4 Hydraulic Controls). The current thrust of WWTF permitting emphasizes meeting effluent concentrations that will protect lakes with reasonable dilution.

Domestic wastewater enters a WWTF with P in excess of 3 mg/L and sometimes as high as 15 mg/L. N levels can exceed 40 mg/L, with values up to 70 mg/L not uncommon. Wastewater treatment in Massachusetts involves primary and secondary treatment and in some cases, tertiary treatment. Primary treatment involves the settling out of suspended solids in sedimentation tanks. Secondary treatment usually involves a biological component to oxidize and convert organic wastes. The two most common methods of secondary treatment are activated sludge reactors (Hanel, 1988) and trickling filters. Effluent is treated with chlorine, ultraviolet light, or ozone before discharge in order to destroy pathogenic organisms (Sundstrom and Klei, 1979). Resulting P concentrations can be as low as 0.3 mg/L, but are more often >1 mg/L and often as high as 3-4 mg/L. N levels of 10-15 mg/L are common, with concern directed toward the fraction of the N load that is present as toxic un-ionized ammonia. Well functioning secondary treatment WWTFs tend to convert nearly all ammonia/ammonium to nitrate.

Advanced waste treatment, or tertiary treatment, usually involves the removal of phosphorus and/or nitrogen. Phosphorus compounds are most often removed by coagulation with chemicals, particularly the addition of alum (see Section 3.5 Phosphorus Precipitation and Inactivation). Occasionally, iron or lime is used. Phosphorus may also be removed by biological processes such as the Anaerobic/Oxic (A/O) process that uses bacteria to remove phosphorus (Bowker and Stensel, 1987). There are many methods to remove nitrogen compounds, including ammonia stripping by air and nitrification-denitrification in biological reactors. Other tertiary treatment methods include adsorption of residual organic and color compounds on activated carbon and the use of reverse osmosis and electrodialysis to remove dissolved solids (Sundstrom and Klei, 1979). Dissolved air flotation (DAF) can also greatly reduce P concentrations, but is more commonly used in drinking water treatment than wastewater situations. Wetland treatment has become popular for nutrient control as a polishing step in WWTFs (Kadlec and Knight, 1996), and some WWTFs are based mainly on biological activity as a mainstay of wastewater treatment.

Achievement of concentrations <1.0 mg/L requires advanced treatment, with attainment of levels as low as 0.02-0.05 mg/L currently sought in several WWTFs, although achievement of levels <0.5 mg/L on a routine basis is rare. The USEPA is reducing effluent concentrations for P as NPDES permits come up for renewal; limits >1.0 mg/L are rarely issued, and targets as low as 0.03 mg/L are being discussed. With a target lake P level of <0.02 mg/L and preferably <0.01 mg/L to minimize algal blooms, WWTF inputs require either greatly enhanced treatment or substantial dilution to avoid eutrophication impacts on lakes. For Massachusetts WWTFs with advanced P removal, monthly mean effluent concentrations ranged from 0.03 to 1.40 mg/L from 1995 to 2001, with annual means (including data only from times with active advanced treatment) ranging from 0.16 to 0.92 mg/L. Where advanced P removal is not practiced, effluent concentrations exceed 1.0 mg/l and are as high as 6.3 mg/L as a monthly mean.

Advanced wastewater treatment has not been implemented as often as desired because of the added cost (J. Dupuis, MDEP, pers. comm., 1995), but has been applied where less stringent treatment has failed to achieve desired results in downstream lakes. Advanced treatment was applied to reduce impacts on Shagawa Lake, Minnesota, as a test project funded by the USEPA. More recently, pilot programs to reduce effluent P to <0.02 mg/L have been conducted, most notably in Syracuse, NY. Advanced wastewater treatment has been used more often in Europe where discharges have been made to a lake, often with results comparable to the diversion of treated effluent (Cooke et al., 1993a). Application in Massachusetts has involved half year (April-October) or full year operation, depending upon the nature of the receiving water; lakes with short detention times have been candidates for half-year advanced treatment requirements.

Note that storm water that is conveyed through any type of drainage system is defined by the USEPA as a point source and subject to NPDES permits. Section 402(p) of the Clean Water Act establishes permit requirements for certain municipal and industrial storm water discharges, and further regulations may apply in the coastal zone (see Coastal Zone Non-Point Pollution Program in Appendix I). The most salient provision of the NPDES program for storm water is the requirement for a Storm Water Pollution Prevention Plan (SWPPP), which is a site- and activity-specific management guide for minimizing impacts on runoff from the site. The emphasis is on prevention of pollution, not treatment or remediation. The SWPPP includes provisions for managing potential pollutants stored or used on site, limiting exposure of potentially polluting activities to precipitation and runoff, and responding to spills, leaks, or other releases. Monitoring provisions are industry-specific and not overly stringent, but the whole process is a major step toward minimizing contamination of runoff and documenting that effort.

In some cases inflows to wastewater treatment plants are combined with urban storm water flow. This is most often a result of underdesign of conveyance systems in the face of expanding user populations, with combined manholes for easy access to both sanitary and storm sewers being the primary point of mixing. This situation leads to excess hydraulic loading to the drainage system and/or WWTF during storms that may result in untreated or incompletely treated wastes being discharged to streams or lakes. Separating these Combined Sewer Systems (CSS) to avoid Combined Sewer Overflow (CSO) has been emphasized by the USEPA and MDEP for about two decades now, and substantial progress has been made.

One less well-known point source that has become a problem in Massachusetts is drinking water treated to comply with anti-corrosion provisions of the federal Safe Drinking Water Act of 1996. The most common chemical used to inhibit corrosion in distribution pipes is calcium phosphate, with concentrations of P in excess of 1 mg/L in many cases and sometimes as high as 5 mg/L, not much different than secondary treated sewage! Blowdown from boilers or hydrants, discharged directly to storm water drainage systems, or leaks from water mains can provide a substantial input of P to downstream lakes. Use of potable water for make-up water in smaller ponds and swimming facilities can actually cause an algal bloom. Alternatives to calcium phosphate, such as a variety of silicates, are more expensive.

3.3.2 Effectiveness

3.3.2.1 Short-Term

Secondary treatment of wastewater is generally ineffective in controlling eutrophication; unless dilution is very high from other water sources, excessive productivity in downstream lakes can be expected. Effectiveness of tertiary treatment of effluent discharged to a lake system will depend on local conditions, especially hydraulic detention time and internal recycling. Where detention time is short and internal recycling is limited, response may be rapid. However, nutrients present in the lake sediments and water column often continue to cause eutrophication and associated algal blooms and plant growth after improved treatment of wastewater, and other techniques may be required to achieve water quality goals.

3.3.2.2 Long-Term

Tertiary point source treatment is often an effective method for nutrient control, and where the discharge from a WWTF is a dominant component of phosphorus loading, it may be an essential step in lake restoration/rehabilitation. Primary treatment removes approximately 10% of total phosphorus (Metcalf & Eddy, 1979). Phosphorus removal by secondary treatment is typically 20-40% of total phosphorus (Sundstrom and Klei, 1979), although higher removal rates are now being achieved fairly routinely. Addition of alum can result in 95% removal of phosphorus during tertiary treatment, but alum is not as effective at removing phosphorus at low ($<5^oC$) temperatures. Additions of lime at pH values near 10 SU can result in 65-80% removal of phosphorus. Chemical removal of phosphorus is best accomplished following the secondary phase of treatment because the phosphorus present at this point is nearly all orthophosphorus, a soluble form of phosphorus that is more easily removed by coagulation reactions (Metcalf & Eddy, 1979). However, simple coagulant addition at key points in the secondary process can reduce the effluent P concentration below 1.0 mg/L without the need for an additional clarifier or filtration step. Advanced P removal in Massachusetts WWTFs has resulted in mean effluent concentrations of 0.16 to 0.92 mg/L in recent years, while WWTFs without treatment or during winter periods of non-treatment have average effluent levels of 1.7 to 3.8 mg/L.

Various biological methods are used for the removal of phosphorus, some of which remove total phosphorus down to 1 mg/L or lower (see various reports in Ramadori, 1987; Bowker and Stensel, 1987). Recent experiments as part of a storm water management program intended to lower P levels in discharges to the Everglades have succeeded in approaching the 0.01 mg/L level through biological control, but the process requires extended detention in large basins and is less reliable than chemical means.

Approximate removal rates for nitrogen from primary and secondary treatment are 5-10% and 10-30%, respectively. Of the many applications for tertiary removal of nitrogen, the most effective include denitrification (70-90% removal), breakpoint chlorination (80-90% removal), selective ion exchange for ammonium (70-95% removal), and ammonia stripping (50-90% removal) (Metcalf & Eddy, 1979).

Removing a nutrient point source is sometimes not enough to reverse the eutrophication process of a nutrient-rich lake. Additional nutrient and plant control measures may be needed. In Lake Shagawa, Minnesota, it was found that even with advanced wastewater treatment, recovery from eutrophication was very slow due to internal nutrient loading. Significant internal nutrient loading was still occurring 16 years after treatment, but summer total phosphorus levels had decreased from a range of 35 to 50 µg/l to a range of 20 to 30 µg/l (Cooke et al., 1993a). The key point is to know the relative importance of internal and external sources and the total reduction necessary to achieve desired conditions.

3.3.3 Impacts to Non-Target Organisms

Adverse impacts to non-target organisms are not expected except possibly for impacts associated with construction of upgraded WWTFs. In addition to removing nutrients, controlling a nutrient point source may reduce oxygen demand and improvements in downstream oxygen concentration may be expected. Long-term improvements in the overall health of the lake would be expected.

One exception to the beneficial nature of point source controls is the potential for nutrient loads to increase, even if the concentration decreases. This can happen if the reduced effluent concentration facilitates greater inflow capacity and more wastewater is passed through the WWTF. Diverting wastewater from local septic systems to a WWTF that discharges to a tributary of a lake could result in more P entering the lake than was delivered from those septic systems. Expansion of the area served by the WWTF can have the same result. Consequently, permit limits need to be expressed as both concentrations and loads to be truly effective.

3.3.4 Impacts to Water Quality

3.3.4.1 Short-Term

An improvement in water quality is expected with tertiary treatment, but may not be observable in the short-term. The rate of water quality improvement will be a function of the magnitude of other sources and the detention time of the lake.

3.3.4.2 Long-Term

Eutrophic conditions and poor water quality may not be reversible by point source nutrient controls (other than diversion, addressed elsewhere) unless tertiary treatment is applied and effluent limits are set well below the common standard of 1 mg/L. It may also be necessary to treat for more than the summer half of the year. Phosphorus levels in wastewater are simply too high and discharges comprise too much of annual low flows to achieve in-lake concentrations <0.02 mg/L without extreme treatment and/or dilution.

Nitrogen removal may also improve water quality, but great care must be taken to avoid lowering nitrogen much more than phosphorus, as a shift to nitrogen limitation will often foster blooms of certain very objectionable blue-green algae. Chlorination, used for disinfection and nitrogen removal, may pose a problem by creating chlorinated organic compounds believed to be health threats. Most of the treatment plants in Massachusetts use chlorination, although some use ultraviolet light or ozone.

3.3.5 Applicability to Saltwater Ponds

Although no literature is available on the use of point source nutrient controls specifically for saltwater ponds, point source controls certainly appear applicable. Saltwater ponds may be limited by nitrogen rather than phosphorus and nutrient testing should be conducted prior to beginning point source nutrient reduction. In cases where nitrogen is the limiting nutrient, the emphasis on treatment processes may change from the more typical phosphorus-focused approach. In addition to nutrient issues, high fecal coliform levels from insufficiently disinfected WWTF effluent or untreated storm water delivered as a point source may threaten shellfish beds and result in closure of shellfishing areas.

3.3.6 Implementation Guidance

3.3.6.1 Key Data Requirements

Data requirements for this type of nutrient control include an accurate nutrient budget including both a measured mass balance and a land-use source analysis. Nutrient budgets should include analysis of all inputs, including internal sources (recycle within the lake, Section 1). Nutrient control should target enough of the load to attain the desired reduction in loading to the lake, with estimates of effectiveness made for lake recovery in terms of total phosphorus levels and Secchi disk transparency. Models of watershed loading and lake response are helpful in this regard, but only mimic reality; the use of several modeling approaches is recommended. Because of the potential for long-term benefits and minimal adverse impacts, point source nutrient control should be encouraged in the watersheds of all lakes. As such discharges must have valid NPDES permits, there is a defined process for setting limits on effluent concentrations and total load that must be followed. Given the high nutrient levels in most point sources, additional techniques may be necessary to achieve desired in-lake conditions.

3.3.6.2 Factors that Favor this Approach

The following considerations are indicative of appropriate application of point source controls for reductions in nutrient concentrations in lakes:

1. A substantial portion of the P and/or N load is associated with point source pollution.
2. Studies have demonstrated the impact of identifiable discharges on the lake.
3. Water associated with point sources is important to lake hydrology.
4. Pollutant removal expected from treatment upgrade has been properly calculated and is achievable.
5. Jurisdiction can be claimed over point sources.

3.3.6.3 Performance Guidelines

Planning and Implementation

Careful consideration should be given to phosphorus removal and to where the effluent is discharged. Ultimately, a desirable target P concentration for point source effluents would be close to 0.02 mg/L, requiring little additional dilution to be acceptable in a downstream lake. This level of treatment has been obtained in some storm water management cases, but is not in use on a full scale basis at any WWTF. Limits of 0.1-0.2 mg/L have the potential to create acceptable downstream conditions with dilution and limited additional inputs, but there is no evidence yet that these limits will stop or reverse eutrophication downstream of the discharge. Additional in-lake methods of nutrient and/or algae control may therefore be necessary.

Because most algal blooms and problems occur during the warmer months, only seasonal phosphorus removal may be required for lakes with short retention times (e.g. $<$2-3 months). However, retention of some portion of the P load and internal recycling suggest that except where detention time is very short (several weeks), this may be an unwise practice.

The key to successful point source control, from the perspective of lake management, is to construct a detailed and reliable nutrient budget and carefully evaluate what any change in load attained by point source control will mean for lake condition. Models of lake behavior in response to nutrient loading are useful in this regard, but as these models are simplifications of reality, the use of multiple modeling approaches is recommended.

Monitoring and Maintenance
Maintenance of WWTFs is an ongoing function of the wastewater authority. Operational errors occur, to be sure, but training and performance of operators is generally high and WWTFs tend to perform to specifications on a fairly regular basis. Those specifications may not have been developed with protection of downstream resources in mind, but most WWTFs meet the assigned permit limits. The primary exception involves facilities with significant infiltration and inflow problems. That is, where storm water can enter the sanitary sewer system, capacity of the WWTF may be overrun during wet weather and treatment effectiveness plummets. Remedial action within the collection system is then needed before treatment reliability can be maintained.

Monitoring nutrient levels in WWTF discharges and in some storm water discharges covered by NPDES is required in some cases, but should not be assumed to be in place. WWTFs with permit limits for specific elements or compounds (ammonium, nitrate, phosphorus) will monitor for those constituents on a weekly to monthly basis in most cases. Over many years, a reliable data base will accrue, but daily information is often lacking. Monitoring the water chemistry of lakes, ponds, or impoundments impacted by a wastewater treatment plant before and after an upgrade is highly recommended. More intensive studies that follow discharges through the system may be very helpful in understanding the magnitude and extent of impacts.

Mitigation
Mitigation for point source discharges is usually a matter of increased treatment to minimize downstream impacts. Unfortunately, the level of treatment for many wastewater discharges is insufficient to avoid substantial impacts, leading to controversy over the siting and management of point source discharges.

3.3.7 Regulations

3.3.7.1 Applicable Statutes

In addition to the standard checklist for projects described in Appendix II, the following specific restrictions and permits include:

- If the discharge may alter or affect a wetland resource, a Notice of Intent must be sent to the Conservation Commission with a copy to the Department of Environmental Protection Regional Office. If the proposed project occurs within an Estimated Habitat of Rare Wildlife in the most recent version of the Natural Heritage Atlas, a copy of the Notice of Intent must be submitted to the Natural Heritage and Endangered Species Program (NHESP) within the MDFG for review (Appendix II). If the proposed project occurs within a Priority Habitat of Rare Species in the most recent version of the Natural Heritage Atlas, the project proponent must submit project plans to the NHESP for an impact determination. An Order of Conditions must be obtained prior to work.
- Any discharge to surface waters in Massachusetts requires a NPDES permit (3.14 CMR 3.0), issued by the USEPA but reviewed by the MDEP. Discharge under this permit does not allow discharge to low flowing or standing waters like lakes and ponds with no outflow (3.14 CMR 4.04). Discharges are generally restricted to large streams or rivers that can handle the

flow of effluent. If the available streams are not large enough, a permit to discharge to groundwater may be granted (Appendix II).

- Simple extensions or connections to existing WWTFs will require a Sewer Extension or Connection Permit (SECP). Discharges to waters may exceed MEPA thresholds, but a MEPA review may be required. For further information on these permits see Appendix II.

In essence, no environmental agency is likely to oppose an effort to reduce the nutrient concentrations and loads discharged from point sources, but there are distinct procedures to be followed and a lengthy review process should be expected. It is more likely that owners of WWTFs will be looking for ways to avoid lowering nutrient levels to meet downstream needs, mainly as a function of cost, and the NPDES process becomes a long-term, iterative process for achieving a desirable discharge limit.

3.3.7.2 Impacts Specific to Wetlands Protection Act

The following overall impact classification is offered as a generalization of impacts, with clarifying notes and caveats as warranted:

1. Protection of public and private water supply – Benefit (surface water quality enhanced).
2. Protection of groundwater supply – Neutral, unless there is a discharge to groundwater, in which case the impact would be a benefit.
3. Flood control - Neutral
4. Storm damage prevention - Neutral
5. Prevention of pollution - Benefit (water quality enhancement).
6. Protection of land containing shellfish - Neutral, but possible benefit through water quality enhancement.
7. Protection of fisheries - Benefit (water quality enhancement), but possible detriment through reduced fertility.
8. Protection of wildlife habitat – Benefit (water quality enhancement), but possible detriment through reduced fertility.

3.3.8 Costs

Advanced wastewater treatment is very expensive to implement. The construction cost in 1973 associated with Lake Shagawa in Ely, Minnesota, was $1.9 million ($6.6 million adjusted to 2000 dollars), and yearly operating costs have averaged about $389,000. Tertiary treatment for WWTF effluent discharged to Lake Zürich, Switzerland, had construction costs of $36 million ($102 million adjusted to 2000 dollars) and yearly operating costs of $1.5 million (Cooke et al., 1993a). Projected costs for operation of the Belchertown wastewater treatment plant are estimated at $300,000 per year (P. Dombrowski, T&B, pers. comm., 1995). Annual operating costs for the Pittsfield wastewater treatment plant, which serves Pittsfield, Hinsdale, Dalton, and North Lenox and handles approximately 17 MGD (million gallons per day) are $1.6 million (T. Landry, City of Pittsfield, pers. comm., 1995). Costs vary with choice of advanced treatment technique, the targeted nutrients, and the desired effluent concentration. As a rough estimating tool, capital cost of tertiary treatment for phosphorus and nitrogen removal will cost at least a million dollars per million gallons treated per day and the operational cost will be at least $100,000 per year per million gallons treated per day. Much higher costs are certainly possible.

3.3.9 Future Research Needs

Less costly methods to reduce N and P in wastewater are needed to make widespread implementation of advanced treatment affordable. A better understanding of aquatic system response to reduced point source loads would aid prediction of management results and enhance planning efforts to reverse eutrophication from point source inputs.

3.3.10 Summary

Wastewater and certain storm water discharges are considered point sources under federal law, which governs the issuance of discharge permits in Massachusetts. Concentrations of N and P in wastewater that has undergone either primary or secondary treatment processes are still at least two orders of magnitude higher than what would be acceptable in most lakes to prevent eutrophication. Concentrations in storm water and in potable water treated to conform with anti-corrosion regulations may also be high enough to warrant major concern. Permit limits under the NPDES program have tended to allow P concentrations of 1 mg/L, and some permits have restricted discharge concentrations only during the growing season. The USEPA is currently reducing discharge limits as permits are renewed.

Considerable treatment or dilution is necessary to reduce inputs to acceptable concentrations, and in many cases the WWTF discharge represents a dominant component of flow during extended dry periods. Mounting evidence from aquatic studies and advancing technology in water treatment indicate that permit limits are too high and that lower concentrations can be achieved. More recently proposed P limits are in the 0.1 to 0.2 mg/L range, with discussion of limits as low as 0.03 mg/L, and with year-round restriction unless detention time is very short. Cost is the major factor preventing implementation of tertiary treatment at more WWTFs.

Modeling should be used to estimate the phosphorus reduction in the lake on a case by case basis. Improvement of conditions will depend upon the portion of the load reduced and the detention time in downstream lakes. The timing of load reductions may also be a factor.

There are no significant environmental disadvantages to upgrading to tertiary treatment. However, care must be taken in permit development to restrict both concentration and load of targeted nutrients, as WWTF expansion could result in an increased nutrient load, even with a lower effluent concentration. Monitoring of the receiving waters and associated impoundments for total phosphorus is recommended before and after upgrading treatment. Monitoring of actual effluent quality will be the responsibility of the wastewater authority, but care should be taken to ensure an appropriate frequency of measurements.

3.4 HYDRAULIC CONTROLS

3.4.1 Overview

There are four basic methods that can be used to take advantage of the flow of water to alter the nutrient concentration in lakes and thus control algal populations. Usually these are used only for algal control because most macrophytes obtain most of their nutrients from the sediments and would not be greatly affected by these methods. The four methods include:

- Diverting nutrient rich wastewater before it reaches the lake.
- Lowering nutrient concentrations in the lake by dilution with low nutrient water.

- Frequently flushing the system with any source of water to minimize the expression of nutrient loads.
- Withdrawal of nutrient-rich water from the bottom of the lake (hypolimnion) before it can interact with surface waters (epilimnion, or the photic zone, where algae grow).

For all four treatments, consideration should be given to alterations in the hydraulic regime of the lake so that inadvertent drawdown or flooding does not result. The nutrient rich water that is diverted, flushed or withdrawn from the lake must be discharged to another location, usually a stream or river somewhere downstream from the lake. Only in the case of dilution is the quality of downstream discharge likely to increase without additional treatment of the discharge.

3.4.2 Diversion

Diverting water from a lake makes sense if the associated nutrient load is undesirable and the loss of the hydrologic load will not have undue negative impacts. Ideally, diversion involves a small amount of water with a large amount of nutrients in it. Diversion is most often practiced in association with wastewater or storm water discharges to lakes with adequate alternative water supplies. It suffers from the philosophical drawback of sending contaminated water elsewhere without addressing the source of nutrients, and may be difficult to permit, but it can be a very effective means of reducing nutrient inputs. Some additional discussion has been provided in conjunction with point source controls (Section 3.3).

3.4.3 Dilution and Flushing

Lake waters that have low concentrations of an essential nutrient are unlikely to exhibit algal blooms. While it is preferable to reduce nutrient loads to the lake, it is possible to lower (dilute) the concentration of nutrients within the lake by adding sufficient quantities of nutrient-poor water from some additional source. High amounts of additional water, whether low in nutrients or not, can also be used to flush algae out of the lake faster than they can reproduce. However, complete flushing is virtually impossible in many lake systems; small, linear impoundments are the primary candidates for such treatment.

Phosphorus is normally the nutrient that limits algal growth. Its concentration in lake water is a function of its concentration in incoming water, the flushing rate or residence time of the lake, and the net amount lost to the sediments as particles settle during water passage through the system. When water low in phosphorus is added to the inflow, the actual phosphorus load will increase, but the mean phosphorus concentration should decrease. The mechanisms associated with this technique are much more complicated than is initially apparent. In-lake concentration could actually increase under some circumstances (Uttormark and Hutchins, 1980), and significant internal phosphorus release can further compromise effectiveness, but dilution has been effective in some cases (Cooke et al. 1993a). A thorough understanding of the phosphorus budget for the lake is necessary to evaluate dilution as a potential algae control method.

Dilution or flushing washes out algal cells, but since the reproductive rate for algae is high (blooms form within days to a few weeks), only extremely high flushing rates will be effective without a significant dilution effect. A flushing rate of 10 to 15% of the lake volume per day is appropriate (Cooke et al., 1993a). Development of reliable water and nutrient budgets are necessary to an evaluation of flushing as an algae control technique.

Very few documented case histories of dilution or flushing exist, in part because additional water is not often available, especially water that is low in nutrients. The best documented case is that of Moses Lake, Washington (Welch and Patmont, 1980; Cooke et al. 1993a), where low-nutrient Columbia River water was diverted through the lake. Water exchange rates of 10 to 20% per day were achieved, algal blooms dramatically decreased, and transparency was markedly improved, illustrating the potential effectiveness of this method.

Outlet structures and downstream channels must be capable of handling the added discharge for this approach to be feasible. Qualitative downstream impacts must also be considered. Water used for dilution or flushing should be carefully monitored prior to use in the lake. Application of this technique is most often limited by the lack of an adequate supply of low nutrient water.

3.4.4 Selective Withdrawal

For recreational lake management, the intent of selective withdrawal is usually to remove the poorest quality water from the lake, which is normally the water at the bottom of the lake unless an intense surface bloom of algae is underway. It is desirable to discharge water at a rate that prevents anoxia near the sediment-water interface, resulting in both improved lake conditions and an acceptable discharge quality. This can be accomplished in impoundments with small hypolimnia and/or large inflows. In most lake management cases, however, selective withdrawal will involve waters of poor quality and treatment may be necessary before discharge downstream.

Where phosphorus has accumulated in the hypolimnion through release from the sediments, selective discharge of hypolimnetic waters prior to fall turnover can reduce effective phosphorus loading. However, unless late summer inflows are substantial, this may result in a considerable drawdown of the lake level. Where a drawdown is planned, selective discharge may increase the benefit. Often an outlet structure must be retrofitted to facilitate selective withdrawal, but the one-time capital cost confers permanent control with minimal operation and maintenance costs.

Nurnberg (1987) reviewed results for 17 lakes with 1 to 10 years of hypolimnetic withdrawal and concluded that reduced epilimnetic phosphorus concentrations did result, presumably leading to lowered algal biomass. However, concerns over summer drawdown, disruption of stratification, and downstream water quality must all be addressed in a successful program.

In some large western reservoirs, hypolimnetic discharges constitute a major outflow and are responsible for maintenance of very productive downstream coldwater fisheries. Aeration or other treatment of discharged water may be necessary, but the removal of phosphorus and other contaminants from the lake can be beneficial. Detailed knowledge of system morphometry, thermal structure, hydrology and phosphorus loading is essential to proper application of this technique.

Selective withdrawal for water supply means locating the intake at the depth where water quality is most advantageous for the intended use. It can be used in any system where vertical water density gradients are sufficiently stable, but is most often applied to more strongly stratified lakes. For potable water use of productive lakes, the choice is often between high algae concentrations in the epilimnion and high iron and/or manganese in the hypolimnion. Intakes located near the thermocline sometimes get both high algae and high metals. A choice of intake depths is preferred, allowing adjustment of intake depth in accordance with the best available water quality. For cooling water supply, cold hypolimnetic withdrawal is preferred, as long as it does not contain high levels of corrosive sulfides.

3.4.5 Effectiveness

Generally these techniques have been shown to be effective where applicable, but the opportunities for these techniques are limited in Massachusetts. The effectiveness of each technique depends mainly on how much the nutrient levels in the lake can be reduced by the method, except in the case of flushing, where algae are physically removed from the lake and nutrient concentration effects are less important.

The effectiveness of each of these methods should be estimated by nutrient budget calculations and simulations of lake response under each treatment method (Section 1). By predicting the nutrient concentrations and detention times resulting from each approach, the potential utility of each can be evaluated.

3.4.5.1 Short-Term Effectiveness of Diversion

The length of time required to observe effects is dependent on such factors as the nutrient input rates and the relative hydraulic detention time of the lake. In lakes with short detention times the response should be quick. In lakes with long detention times the response may be delayed. Additionally, lakes with high internal nutrient recycling rates may have a slow recovery (Cooke et al., 1993a). Diversion has worked successfully to recover lake quality in cases where external loading dominates the nutrient cycle, but is not usually a fast process.

3.4.5.2 Long-Term Effectiveness of Diversion

The long-term response of a lake to diversion is usually favorable if the diversion is effective at reducing nutrient concentrations. Results from several diversion projects have shown that there is a high probability for lake recovery. The Lake Washington example is probably the most well documented case where the diversion of Metropolitan Seattle secondary treated domestic sewage from Lake Washington to Puget Sound resulted in a dramatic improvement in water quality over time. In this case 88% of the external phosphorus loading was diverted and the TP in the lake declined from about 64 μg/l to about 17 μg/l after five years (Edmondson, 1977). Lake Washington has maintained desirable water quality for many years following diversion in 1967 (Edmondson and Lehman, 1981).

Lake Sammamish (Issaquah, WA) showed a very slow response to wastewater diversion, with little response in the first 7 years even though the flushing rate was similar to Lake Washington. This was attributed to high internal loading of phosphorus, which was reduced in later years as the hypolimnetic oxygen deficit rate improved and the more oxic conditions inhibited phosphorus release from the sediments (Welch et al., 1984).

Lake Norrviken is another example of a diversion project that required a longer time period to recover than Lake Washington and recovered to a lesser degree. The maximum concentration in the lake at turnover declined from approximately 450 μg/l to 150 μg/l and summer levels of total phosphorus decreased from 263 to 174 μg/l between 1970 and 1979. This change resulted in a reduction in chlorophyll a (chl a) and an improvement in transparency. Although Norrviken is considered a success by many because of the resultant transparency, the lake is still eutrophic (Ahlgren, 1978).

3.4.5.3 Short-Term Effectiveness of Dilution and Flushing

The effects of dilution and flushing can reduce algal abundance in two ways. The first is by direct dilution of nutrient concentrations in the lake by the addition of low nutrient water, resulting in nutrient limitation of algal growth. The second is by the physical removal of algae in the discharge water. In the latter case, it is possible to reduce algal abundance even if the nutrient level in the inflow is higher than the lake's nutrient level, but only if the cells are flushed out of the lake at a rapid rate. How quickly the nutrients can be diluted or the algae removed depends on the flushing rate. One might reasonably assume that if these techniques are going to be effective, the results will be detectable within a few weeks of initiation.

3.4.5.4 Long-Term Effectiveness of Dilution and Flushing

As previously mentioned, few studies are available on dilution and flushing as a treatment for lakes. These treatments are expected to be effective for as long as they are applied. It should be noted that in Green Lake, Washington, effectiveness declined after initial success, due to the reduction in inflow dilution water. The cost of using city water to dilute the lake was simply too expensive to continue for a long time period (Cooke et al., 1993a).

3.4.5.5 Short-Term Effectiveness of Selective Withdrawal

Hypolimnetic waters in eutrophic lakes are often anoxic. The lack of oxygen promotes the release of phosphorus from the sediment, resulting in high concentrations that can be entrained and transferred to the epilimnion during a later mixing event. If this hypolimnetic water is removed and replaced by epilimnetic water that is higher in oxygen, the periods of anoxia should decrease and the rate of sediment release of phosphorus is expected to be reduced. This reduction in internal phosphorus inputs, combined with the flushing of nutrient-rich hypolimnetic water out of the system, is expected to result in decreases in epilimnetic nutrient concentrations. The effectiveness of this treatment is not expected to be significant in the short-term (weeks to months) however, because stratified lakes usually do not mix significant amounts nutrients from the hypolimnion to the epilimnion until fall turnover occurs. As a result, reduction of epilimnetic concentrations may not be observed until the following spring.

3.4.5.6 Long-Term Effectiveness of Hypolimnetic Withdrawal

Hypolimnetic withdrawal is one technique that appears to become more effective the longer it is used. Nürnberg (1987) found that several years are generally needed to show significant improvements in epilimnetic TP following the initiation of hypolimnetic withdrawal. In a review of case studies of hypolimnetic withdrawal, Cooke et al. (1993) reported that epilimnetic P decreased in 8 of 12 lakes for which there were one or more years of data. Lake Wononscopomuc and Lake Waramaug in Connecticut showed some improvement in water quality following hypolimnetic withdrawal, but only Lake Wononscopomuc showed a significant decrease in epilimnetic P, while Lake Waramaug showed no significant trend (Nürnberg et al., 1987). Later calculations revealed that the P export via the withdrawal pipe in Lake Waramaug decreased the internal load by only 10-20% due to sub-optimum pipe placement (Nürnberg et al., 1987).

3.4.6 Impacts to Non-Target Organisms

3.4.6.1 Short-Term

All of these techniques involve discharging more nutrients ~~in~~ to another location, usually downstream of the lake. It is expected that there will be improved conditions in the lake for most non-target organisms, but this may not be the case downstream or wherever diverted flows are discharged. Most organisms will tolerate minor short-term fluctuations in flow and water quality, so short-term impacts are not expected to be severe. An exception would be the possible short-term impacts associated with a discharge of low oxygen water that dominates downstream hydrology and suffocates aquatic life. For this reason, some hypolimnetic discharges from large reservoirs are aerated by spraying the water into the air as it is discharged. The hypolimnetic discharge from Lake Waramaug was aerated in a raceway before discharge and treated to reduce selected contaminant levels. In the case of selective withdrawal, caution must be exercised not to induce summer drawdown. Should such a drawdown occur, the potential impacts associated with drawdown (Section 4.2) must be considered.

3.4.6.2 Long-Term

Long-term impacts in the lake are likely to be positive, based on reduced nutrient availability. Long-term downstream impacts will be a function of the duration, magnitude and quality of discharges, plus the sensitivity of downstream biota. For Lake Washington, the wastewater entering from eleven small secondary treatment plants located around the lake was collected, treated and eventually discharged at depth into Puget Sound where it was assumed to have fewer impacts (Edmondson and Lehman, 1981). This may not be the case in all systems, however, and downstream studies appear necessary before attempting any of these methods. In most cases it is advisable to aerate the discharge water in order to raise the dissolved oxygen content, remove any toxic hydrogen sulfide gas, and reduce concentrations of ammonium, iron and other metals that may otherwise exceed regulatory limits for discharges (Nürnberg et al., 1987).

Stream flow can have an impact on fish populations as different species habitats are dictated by depth, current velocity and area, as well as stability of flow (Lewis, 1969; Bain et al., 1988). For example, a group of small fish species in the Deerfield River, Massachusetts and the West River in Vermont are restricted to a microhabitat of shallow, slow waters along stream margins (Bain et al., 1988). Alteration of flow and water quality may affect such assemblages, and similar impacts on invertebrate communities might be expected.

Increased turbidity resulting from increased flows may also pose a potential impact in the receiving waters. Some impacts are to be expected during the placement of pipes, particularly during the construction period, but this depends on the scale of the diversion and the distance to reach the discharge location.

For hypolimnetic withdrawal there may be entrainment of small organisms, or impingement on the screens of the intake pipe, assuming the organisms can withstand the anticipated low oxygen levels in the hypolimnion where the intake pipe would be located.

3.4.7 Impacts to Water Quality

3.4.7.1 Short-Term

For properly applied dilution and diversion, water quality should improve rapidly the lake. Delays may result from long detention time or excessive internal loading, but these techniques may not be the best choices in such circumstances. Short-term in-lake improvement is not expected from hypolimnetic withdrawal. Chemical water quality may not change appreciably for flushing strategies, although reduced algal abundance may induce some changes (e.g., suspended solids, pH). Negative impacts to downstream water quality (or wherever discharges are diverted) may be rapidly manifest unless the discharged water is treated or otherwise shown to have acceptable quality. Impacts from elevated flow should not occur in a properly planned program, as flows should be kept within the natural range for downstream channels, but some potential exists for flushing as a consequence of higher overall flows. In the case of dilution, if the dilution water is from deep water wells, the water may be low in dissolved oxygen and have high metals or sulfide content, which may adversely impact some aspects of water quality (HWH, 1990a).

3.4.7.2 Long-Term

Long-term impacts on water quality are similar to the short-term impacts described above. One notable exception is that with hypolimnetic withdrawal treatments, improvements in lake water quality are sometimes delayed for several years (Nürnberg, 1987). Under ideal conditions, hypolimnetic withdrawal can maintain improved conditions in the hypolimnion, removing water at a rate fast enough to prevent anoxia. Increased temperature or destratification could result, however, with variable impacts on water quality and biota.

3.4.7.3 Applicability to Saltwater Ponds

These techniques could be applied to saltwater ponds although there are no reports of saltwater application in the literature. In theory, diversion treatments could be applied to divert wastewater inputs to the ocean, although the diversion of wastewater anywhere is a difficult proposition. In many cases, nitrogen rather than phosphorus may be the limiting element in saltwater ponds, but this is less of a factor in these techniques than many others. Sea water itself may be used to dilute and flush eutrophic saltwater ponds isolated from the ocean by barrier sand dunes by dredging open a new connection to the sea (Section 3.7, Dredging). As stated in the Division of Wetlands and Waterways Policy 91-2, this will only be permitted if the purpose is to maintain an existing or historically viable marine fishery and steps are taken to minimize adverse impacts associated with the project. Hypolimnetic waters in eutrophic saltwater ponds may have high sulfide concentrations that may cause toxicity problems and require treatment prior to discharge.

3.4.8 Implementation Guidance

3.4.8.1 Key Data Requirements

Data requirements include accurate hydrologic and nutrient budgets, an assessment of probable in-lake effects, and an evaluation of downstream impacts. In most cases where these techniques were ineffective, the cause was inaccurate nutrient budgets that overestimated the treated source and underestimated other sources of nutrients. If the major input of nutrients to the lake is from a point source, diversion of this source should be effective if such diversion is feasible. If nutrient-poor water is available in sufficiently large quantities, then dilution may be effective. If enough water is available to reduce detention to <2 weeks, flushing may be effective. If the nutrient budget reveals that much of the nutrient load is being recycled from nutrient-rich hypolimnetic waters, then hypolimnetic withdrawal may be effective. For all these methods calculations should be presented to show the volumes of water and nutrient concentrations involved, and how the changes in lake discharge may affect habitat and downstream flow rates. Estimates of effectiveness should be made for lake recovery in terms of total phosphorus levels and Secchi disk transparency.

3.4.8.2 Factors that Favor this Approach

The following considerations are indicative of appropriate application of hydraulic controls for reductions in nutrient concentrations and control of algae in lakes:

1. A substantial portion of the P and/or N load is associated with sources that can be diverted, diluted or preferentially discharged.
2. Studies have demonstrated the impact of identifiable sources (e.g., a discharge, hypolimnetic load) on the lake.
3. Water associated with sources to be diverted or discharged is not important to lake hydrology; water level fluctuation will not differ greatly from pre-treatment conditions.
4. Adequate water of a suitable quality is available for dilution or flushing.
5. Downstream problems with water quantity or quality will not be caused.
6. Actual reduction in nutrient inputs from identifiable sources is not practical, either for technical or jurisdictional reasons.

3.4.8.3 Performance Guidelines

<u>Planning and Implementation</u>

It is imperative that reliable nutrient and water budgets be developed to obtain a reasonable prediction of improvements in nutrient content in the lake before any of these techniques are used. Seasonal application (e.g., during the late spring and summer) may be sufficient to reduce nutrient and algae concentrations in the lake.

Effects of diversions on the lake water budget should be considered in the planning stage. BMPs can be employed to limit environmental impacts associated with any necessary construction. The diverted water (typically wastewater) can be treated more thoroughly prior to discharge to minimize impacts on the discharge site.

Acquiring and controlling the amount of water used to dilute and flush the lake is the primary consideration for these techniques. An adequate source of water must remain to keep the lake water budget in balance, in order to avoid an unintended water level decrease. Hypolimnetic withdrawal is more often applied in the fall to cause a drawdown, accomplishing two goals at once. It is possible to remove sufficient cold water from the hypolimnion such that the lake could become thermally unstable and destratify, perhaps eliminating cold water fisheries. Generally, this has not been observed, as the hypolimnion remains somewhat cooler than the epilimnion in most cases. If this is suspected to be a problem however, it can be counteracted by input of cold stream or well water directly into the hypolimnion in conjunction with hypolimnetic withdrawal (see Figure 7-1 in Cooke et al., 1993a). In the case of Lake Waramaug, the depth of the intake pipe in the hypolimnion had to be raised during summer stratification because the concentration of nutrients, iron and hydrogen sulfide were too high to be discharged. Unfortunately, this also limited the effectiveness of the treatment for removing nutrients (Nürnberg et al., 1987).

Monitoring and Maintenance
Maintenance requirements may be high for systems that involve large amounts of pipe, canals and pumps, but these techniques are not typically implemented if maintenance needs are high. Very little maintenance is required for hypolimnetic withdrawal unless treatment is necessary before discharge. Diversion will normally involve additional piping, but gravity flow systems with minimal maintenance needs are strongly preferred. Dilution or flushing water may be piped and/or pumped, with periodic maintenance needed, but successful systems are as simple as possible.

All treatments require monitoring to make sure that excessive amounts of water are not removed or added. Monitoring would include periodic measurement of discharge volumes and water quality of both the discharge water and the receiving waters. Such monitoring should include nutrients, dissolved oxygen, iron, sulfide, temperature, pH and turbidity to insure that no adverse impacts are occurring. If pollutant content is high, further monitoring of downstream conditions may be warranted. In the case of hypolimnetic withdrawal, periodic measurements of hypolimnetic nutrient content, oxygen content and stability of the hypolimnion are recommended.

Mitigation
Undesired effects are most often mitigated by simply ceasing the hydraulic control. Additional mitigative measures should be considered on a case by case basis. Mitigative measures for hypolimnetic withdrawal include aeration and treatment of the water prior to discharge.

3.4.9 Regulations

3.4.9.1 Applicable Statutes

These methods will involve a Notice of Intent being sent to the Conservation Commission with a copy to the Department of Environmental Protection Regional Office. If the proposed project occurs within an Estimated Habitat of Rare Wildlife in the most recent version of the Natural Heritage Atlas, a copy of the Notice of Intent must be submitted to the Natural Heritage and Endangered Species Program (NHESP) within the MDFG for review (Appendix II). If the proposed project occurs within a Priority Habitat of Rare Species in the most recent version of the Natural Heritage Atlas, the project proponent must submit project plans to the NHESP for an impact determination. An Order of Conditions must be obtained prior to work.

A Chapter 91 Permit may be required for structural alterations in Great Ponds. For any alteration involving a dam, a MDCR Office of Dam Safety Permit may be required (Appendix II). Withdrawal, discharge, or diversion of water in excess of 100,000 gpd may require a permit under the Water Management Act (Appendix II). Any of these techniques may also require a 401 WQ permit, but jurisdiction of the MDEP will depend upon which other permits are required and funding sources. Approval from the Army Corps of Engineers ACOE (Appendix II) is not typically required for these techniques. A MEPA review may be required if certain thresholds are exceeded (Appendix II). Diversion and possibly flushing or withdrawal projects will be subject to NPDES permitting; water to be diverted will most likely already be subject to NPDES, while the need for NPDES permits for flushing or withdrawal will depend upon the quality of water involved.

3.4.9.2 Impacts Specific to Wetlands Protection Act

The following overall impact classification is offered as a generalization of impacts, with clarifying notes and caveats as warranted:

1. Protection of public and private water supply – Variable (depends on location of supply relative to discharge and detention time).
2. Protection of groundwater supply – Neutral, unless there is a discharge to groundwater or a major withdrawal for dilution/flushing, in which case the impact could be detrimental.
3. Flood control - Neutral (added flow must remain within tolerance limits for lake and downstream receiving waters)
4. Storm damage prevention – Neutral (added flow must remain within tolerance limits for lake and downstream receiving waters)
5. Prevention of pollution - Benefit in the lake (water quality enhancement), but possible detriment downstream (possible poor quality discharges).
6. Protection of land containing shellfish - Possible benefit through water quality enhancement in the lake and possible detriment with any downstream water quality degradation.
7. Protection of fisheries - Benefit (water quality enhancement), but possible detriment through reduced fertility and possible detriment downstream with any water quality degradation.
8. Protection of wildlife habitat – Benefit (water quality enhancement), but possible detriment through reduced fertility.

3.4.10 Costs

All of these techniques usually have potentially high capital costs due to construction, with variable maintenance costs, (high for pumping, low for gravity flow). Cost factors to consider include the location and relative elevation of the water source or discharge point in relation to the lake and the discharge site. Costs will depend on the volumes of water to be moved and the distances involved. If a favorable drop in elevation is not present, pumping costs may be substantial. If dilution or flushing water must be purchased, costs will escalate.

3.4.10.1 Diversion

The cost of diversion varies greatly from each site. The cost is primarily based on the required distance for transport and associated construction costs. If the water must be treated prior to discharge, that cost should also be included. Estimates for diversion of various wastewater discharges in Massachusetts (Belchertown WWTF/Forge Pond and Spencer WWTF/Quaboag Pond) have exceeded $5 million; these diversions were not implemented, in favor of improved treatment.

3.4.10.2 Dilution and Flushing

The cost of dilution and flushing varies mainly with the volume and availability of water. The primary costs for Moses Lake were $497,000 (2000 dollars) according to Cooke et al. (1993). As mentioned above, the costs for flushing Green Lake became too expensive; the use of city water was projected to cost $17.7 million dollars over 20 years (Cooke et al., 1993a). If a nearby upstream source of clean water could be diverted to a lake by gravity, or if a short canal can be constructed to provide a connection to a larger stream or river, actual water costs may be considerably less. However, the cost of permitting and constructing the connection to deliver the water may be substantial. Wagner (2001) suggests a cost of $500-2500/acre for application of these techniques, inclusive of permitting and monitoring, when a source of water is readily available. Costs may rise to $5,000-25,000/acre if water is purchased, piped and/or pumped.

3.4.10.3 Hypolimnetic Withdrawal

Installation costs for withdrawal pipes typically range between $3,000 and $45,000, although in Lake Ballinger (Seattle, WA) the cost was $304,000, due to additional construction of a stream water inlet diversion to the hypolimnion (HWH, 1990a; Cooke et al., 1993a). Costs for treating withdrawn water prior to discharge could be substantial, but in most cases where this technique has been applied, treatment has consisted mainly of aeration by passive means at limited capital and minimal operational cost. Wagner (2001) suggests a cost of <$100 per acre where structures are in place and no major downstream impacts are expected. The cost may rise to $1000-3000/acre where structural alterations and/or treatment of discharged water become necessary.

3.4.11 Future Research Needs

These techniques have been applied on only a very limited basis in Massachusetts, and indeed elsewhere as well. More experience with implementing hydraulic controls and mitigating possible negative impacts is needed.

3.4.12 Summary

Diversion and dilution can be very effective in reducing nutrient levels and resultant algal concentrations, but it is rare to find an approved disposal site or source of clean water required for successful application. Flushing will not typically reduce nutrient levels, but may reduce algal density if detention time is lowered to <2 weeks. Hypolimnetic withdrawal may be effective in lakes with nutrient-rich bottom waters, and may even eliminate poor water quality at the bottom. However, issues with poor downstream water quality may require treatment before discharge and maintenance of acceptable hypolimnetic water quality requires a high removal rate during summer. Where drawdown is not intended or tolerable, a compensatory increase in inflow will be needed to allow adequate withdrawal. As summer flows are often low, hypolimnetic withdrawal is normally applied in conjunction with fall drawdown.

In lakes with high nutrient inputs from accumulated sediments, the effectiveness may be somewhat less and somewhat delayed. In general, researchers have been fairly successful at predicting the response of a lake to these types of treatments. The major disadvantages are potentially high capital costs to facilitate these techniques and the potential for downstream impacts. Maintenance of lake level during summer may restrict the utility of diversion and withdrawal. Need for a major water source usually restricts dilution and flushing.

3.5 PHOSPHORUS PRECIPITATION AND INACTIVATION

3.5.1 Overview

The release of phosphorus stored in lake sediments can be so extensive in some lakes and reservoirs that algal blooms persist even after incoming phosphorus has been significantly lowered. Phosphorus precipitation by chemical complexing removes phosphorus from the water column and can control algal abundance until the phosphorus supply is replenished. Phosphorus inactivation typically involves some amount of phosphorus precipitation, but aims to achieve long-term control of phosphorus release from lake sediments by adding as much phosphorus binder to the lake as possible within the limits dictated by environmental safety. It is essentially an "anti-fertilizer" addition.

This technique is most effective after nutrient loading from the watershed is sufficiently reduced, as it acts only on existing phosphorus reserves, not new ones added post-treatment. In-lake treatments are used when phosphorus budget studies of the lake indicate that the primary source of the phosphorus is internal (i.e., recycled from lake sediments). Such techniques have been used for several decades, but we are still learning how to best apply them. Such nutrient control generally does not reduce macrophyte abundance (Mesner and Narf, 1987). On the contrary, the increased light penetration may cause an increase in macrophyte populations. Macrophyte control techniques may be used in combination with phosphorus precipitation and inactivation (Morency and Belnick, 1987; Cooke et al., 1993a).

The three most common treatments for lakes employ salts of aluminum, iron, or calcium compounds. Nitrate treatments are very rare and are used to enhance phosphorus binding to natural iron oxides in sediments. For the aluminum, iron and calcium treatments, the typical compounds used include aluminum sulfate ($Al_2(SO_4)_3\ xH_2O$), sodium aluminate ($Na_2Al_2O_4\ xH_2O$), iron as ferric chloride ($FeCl_3$) or ferric sulfate ($Fe_2(SO_4)_3$), and calcium as lime ($Ca(OH)_2$) or calcium carbonate ($CaCO_3$). Additional forms of aluminum are becoming more common, but these are the normally encountered phosphorus inactivators.

These are applied to the surface or subsurface, in either solid or liquid form, normally from a boat or barge. These compounds dissolve and form hydroxides, $Al(OH)_3$, $Fe(OH)_3$, or in the case of calcium, carbonates such as calcite ($CaCO_3$). These minerals form a floc that can remove particulates, including algae, from the water column within minutes to hours and precipitate reactive phosphates. Because aluminum and iron added as sulfates or chlorides dissolve to form acid anions along with the formation of the desired hydroxide precipitates, the pH will tend to decrease in low alkalinity waters unless basic salts such as sodium aluminate or lime are also added. Conversely, calcium is usually added as carbonates or hydroxides that tend to raise pH. This is especially true for hydroxides that have high solubility (Stumm and Morgan, 1981).

In addition to precipitation from the water column, the floc can inactivate phosphorus in the sediments. The floc settles on the sediment surface, gradually mixes with the upper few centimeters of sediment, reacts with available phosphorus, and prevents the release of phosphorus back into the water column. The resulting nutrient limitation in the surface waters prevents algal blooms from forming.

The various floc minerals behave very differently under high or low dissolved oxygen and they also differ in their response to changes in pH. Because of its ability to continue to bind phosphorus under the widest range of pH and oxygen levels, aluminum is usually the preferred phosphorus inactivator. Other binders are applied under specific conditions that favor their use, but not as commonly as aluminum.

3.5.2 Use of Aluminum Compounds

Aluminum has been widely used for phosphorus inactivation, mostly as aluminum sulfate (alum) and sometimes as sodium aluminate (aluminate), as it binds phosphorus well under a wide range of conditions, including anoxia. However, concentrations of reactive aluminum (AL^{+3}) are strongly influenced by pH. Aluminum is toxic to fish at levels of 100 to 200 µg/L at pH of 4.5 to 5.5 SU, typically via gill membranes (Baker, 1982). The safe level of dissolved aluminum is considered to be 50 µg/L (Kennedy and Cooke, 1982), but this is not a sharp threshold.

Common application rates are in the range of 5,000 to 40,000 µg Al/L (5 to 40 mg/L, see review in Cooke et al., 1993b), but nearly all of this forms an insoluble precipitate (called "floc") and is quickly removed from the water column. A pH of between 6.0 and 7.5 virtually ensures that the 50 µg/L limit will not be reached, although it was thought that a pH of up to 8.5 could be tolerated until recently (ENSR, 2001b). Yet aluminum sulfate addition can reduce the pH well below a pH of 6.0 in poorly buffered waters, and overbuffering can raise the pH above this safe range.

Sodium aluminate, which raises the pH while providing more aluminum, has been successfully used in combination with aluminum sulfate (Cooke et al., 1993b), but has also caused fishkills in two New England lakes (Hamblin Pond in 1995 and Lake Pocotopaug in 2000) due to an improperly low ratio of alum to aluminate. It is also possible to add other buffering agents to the lake prior to aluminum sulfate addition, such as lime, sodium hydroxide, or sodium carbonate. The key is to balance the acids and bases to cause minimal change in pH; fishkills have resulted from a failure to do this. Jar testing is usually employed to evaluate the best ratio of acid and base compounds, but results of lab tests can sometimes be misleading. A fine level of detail is needed to arrive at the correct ratio. Field tests with careful monitoring appear in order for larger scale projects. A volumetric ratio of aluminum sulfate to sodium aluminate of 2:1 is expected to cause no change in system pH. Maintenance of the ambient pH is an appropriate goal, unless the pH is especially high as a consequence of excessive algal photosynthesis.

In practice, aluminum compounds are added to the water and colloidal aggregates of aluminum hydroxide are formed. These aggregates rapidly grow into a visible, brownish white floc, a precipitate that settles to the sediments over the following hours, carrying sorbed phosphorus and bits of organic and inorganic particulate matter in the floc. After the floc settles to the sediment surface, the water will usually be very clear. If enough alum is added, a layer of 1 to 2 inches of aluminum hydroxide floc will cover the sediments, mix with the upper few centimeters, and significantly retard the release of phosphorus into the water column as an internal load.

Aluminum sulfate is often applied near the thermocline depth (even before stratification) in deep lakes, providing a precautionary epilimnetic refuge for fish and zooplankton that could be affected by dissolved reactive aluminum. Application near the surface provides no refuge, but strips phosphorus from the whole water column and provides more immediate removal of phosphorus. Application methods include modified harvesting equipment, outfitted pontoon boats, and specially designed barges made for this purpose.

Good candidate lakes for this procedure are those that have had external nutrient loads reduced to an acceptable level and have been shown, through a diagnostic-feasibility study, to have a high internal phosphorus load (release from sediment). High natural alkalinity is also desirable to provide buffering capacity. Highly flushed impoundments are usually not good candidates because of an inability to limit phosphorus inputs. Treatment of lakes with low doses of alum may effectively remove phosphorus from the water column, but may be inadequate to provide long-term control of phosphorus release from lake sediments and will not affect later inputs of phosphorus from the watershed. High doses are needed to effectively bind phosphorus in the upper few inches of sediment and retard release (Rydin and Welch, 1998); high initial alkalinity, added buffering capacity, or sequential dosing are needed to control water column pH in such treatments.

Nutrient inactivation has received increasing attention over the last decade as long lasting results have been demonstrated in multiple projects, especially those employing aluminum compounds (Welch and Cooke, 1999). Annabessacook Lake in Maine suffered algal blooms for 40 years prior to the 1978 treatment with aluminum sulfate and sodium aluminate (Cooke et al., 1993b). Low buffering capacity necessitated the use of sodium aluminate. A 65% decrease in internal phosphorus loading was achieved, blue-green algae blooms were eliminated, and conditions have remained much improved for nearly 20 years. Similarly impressive results have been obtained in Cochnewagon Lakes in Maine using the two aluminum compounds together (Connor and Martin, 1989a; Monagle, Cobbossee Watershed District, pers. comm., 1995).

Kezar Lake in New Hampshire was treated with aluminum sulfate and sodium aluminate in 1984 after a wastewater treatment facility discharge was diverted from the lake. Both algal blooms and oxygen demand were depressed for several years, but began to rise more quickly than expected (Connor and Martin, 1989a; 1989b). Additional controls on external loads (wetland treatment of inflow) reversed this trend and conditions have remained markedly improved over pre-treatment conditions for almost 15 years. No adverse impacts on fish or benthic fauna have been observed despite careful monitoring.

Aluminum sulfate and sodium aluminate were again employed with great success at Lake Morey, Vermont (Smeltzer, 1990). A pretreatment average spring total phosphorus concentration of 37 µg/L was reduced to 9 µg/L after treatment in late spring of 1987. Although epilimnetic phosphorus levels have varied since then, the pretreatment levels have not yet been approached. Hypolimnetic phosphorus concentrations have not exceeded 50 µg/L. Oxygen levels increased below the epilimnion, with as much as 10 vertical feet of suitable trout habitat reclaimed. Some adverse effects of the treatment on benthic invertebrates and yellow perch were observed immediately after treatment (e.g., smothering of some invertebrates by the floc layer and poor growth by yellow perch for a season), but these proved to be transient phenomena and conditions have been acceptable and stable for over a decade (Smeltzer et al., 1999).

Phosphorus inactivation has also been successful in some shallow lakes (Welch et al., 1988; Gibbons, 1992; Welch and Schrieve, 1994), but has been unsuccessful in cases where the external loads have not been controlled prior to inactivation (Barko et al., 1990; Welch and Cooke, 1999). Successful dose rates have ranged from 3 to 30 g Al/m^3 (15 to 50 g Al/m^2) with pH levels remaining between 6.0 and 8.0 SU.

Considerable advances in dose determination and treatment approaches have been made in the last few years, and continued advances are expected. Low doses (1-5 mg Al/L) can be used to strip phosphorus out of the water column with limited effects on pH or other water quality variables, even in many poorly buffered waters. Mixing with aeration systems can increase treatment efficiency and lower the necessary dose. At the other extreme, determination of available phosphorus in sediments has revealed that higher doses (often in excess of 100 g/m^2) than normally applied are needed to thoroughly inactivate phosphorus reserves and maximize treatment longevity (Rydin and Welch, 1998; 1999). Doses around 10 mg Al/L are typically applied to storm water discharges, which can be automatically dosed in response to storm flows (Harper et al., 1999). Current efforts in storm water management with alum focus on capturing the floc in detention areas prior to discharge to the lake or stream.

Areal doses (g/m^2) convert to volumetric doses (g/m^3 or mg/L) simply by dividing the areal dose by the water depth in meters. However, this means that an areal dose of 50 g/m^2 applied to a 10 ft (3 m) deep section of lake will yield a volumetric dose of 16.7 g/m^3 if added all at once. Without careful buffering, doses of >5-10 g/m^3 have been associated with fishkills, so such high doses require one or more mitigative measures. In the re-worked Lake Pocotopaug treatment of 2001, the alum:aluminate ratio was maintained at 2:1, the dose was split in half (25 g/m^2 or 4 g/m^3 applied twice), and a minimum one-day lag time was allotted between treatments of any one area (ENSR, 2001b). In the 2001 Ashumet Lake treatment, alum:aluminate ratio was also carefully controlled at 2:1, application was made below the thermocline (35 ft, where it was anoxic and no fish or invertebrate life was expected), a pilot treatment was conducted with several days of monitoring afterward, and extensive monitoring was conducted during treatment (ENSR, 2002e). All of these precautions may not be necessary in any one treatment, and greatly increase the cost, but they have facilitated clear demonstration of the success of inactivation without toxic impacts.

3.5.3 Use of Iron Compounds

Iron works very much like aluminum, forming hydroxides that bind phosphorus and make it unavailable for algal uptake. Iron is more common naturally than aluminum, and is abundant in most Massachusetts waters. However, the results of treatment with iron salts are very sensitive to dissolved oxygen levels. Under oxic conditions the ferric hydroxide floc is stable at normal pH conditions (pH>5). Under anoxic conditions, however, the iron in ferric hydroxide is reduced to soluble ferrous iron (Fe^{+2}) and the floc dissolves, releasing the adsorbed phosphorus (Mortimer, 1941; 1942). Therefore, while iron acts as a natural binder in well-oxygenated systems, loss of oxygen in eutrophic lakes may disrupt this natural phosphorus inactivation process.

Inactivation of phosphorus by iron will become very ineffective where anoxia is so strong that sulfate reduction occurs. In such cases, iron is preferentially bound by sulfides released as hydrogen sulfide when oxygen is removed from sulfates by anaerobic bacteria. Iron sulfides are minimally soluble and precipitate out of the water column, further disrupting the natural process of iron-mediated phosphorus control. If oxygen is restored to the system, natural levels of iron may be adequate to bind available phosphorus. Where iron concentrations are inadequate, iron can be added to the system. Consequently, iron is only used in well-aerated systems with naturally low iron levels, but may be the inactivator of choice as a supplement to an aeration system. Iron is used in conjunction with aeration in the water supply of St. Paul, MN, and appears to be successful (Walker et al., 1989).

Iron is generally not toxic at levels applied to lakes but direct information on effects on non-target organisms is lacking. No long-term impacts are reported. Impacts to water quality are expected to be beneficial. Excessive iron can cause rust stains in laundry and sinks, but the added iron is expected to rapidly precipitate out of solution. Excess iron in a water supply may be an issue, as taste and aesthetic aspects of water delivered to customers are important. However, in recreational lakes such concerns are minimal, and iron can provide control of phosphorus where oxygen levels are adequate.

3.5.4 Use of Calcium Compounds

The stability of calcite is highly sensitive to pH, calcium, and carbonate concentrations. Consequently, treatment with calcium is effective only if pH is maintained at a relatively high level (8 SU or above). Such pH levels are found naturally only in the Berkshire region (Mattson et al., 1992), and elevating the pH by chemical addition to facilitate calcium effectiveness may have many adverse impacts on natural systems adjusted to lower pH. Calcium is more commonly used in alkaline lake regions, such as Alberta, Canada, and has not been applied in Massachusetts or the northeastern USA except on a pilot basis (ENSR, 1997a). A general discussion and graph of calcite stability is presented in Section 5.3 of Stumm and Morgan (1981).

Calcium treatments have been effective in reducing algae and total phosphorus in extremely eutrophic lakes that are also hardwater lakes. For example, Halfmoon Lake in Alberta (101 acres, pH of 8.9-9.2 SU, alkalinity of 139 mg/L) was treated with 188 metric tons of $Ca(OH)_2$ and 58 metric tons of $CaCO_3$ over two years for a rate of 120 g Ca/m^2 in 1988 and 182 g Ca/m^2 in 1989 (Babin et al., 1994). This treatment reduced the total phosphorus and chlorophyll a by an estimated 54 and 24 percent, respectively, and sediment phosphorus loading was also reduced. It should be noted that pretreatment concentrations of total phosphorus and chlorophyll a were very high, 124 μg/l and 50 μg/l , respectively, and even with the reduction the post-treatment concentrations were still rather high.

Application involves spreading a powdered form or a slurry made from the powder. Most applications have been made at the surface with spray or gravity feed systems.

3.5.5 Use of Nitrate Compounds

Nitrate treatments such as $Ca(NO_3)_2$, known also by the trade name Riplox, are included here, but nitrates neither precipitate nor inactivate phosphorus directly. Nitrates are injected directly into the surface sediments as a sediment oxidation treatment, which in this case refers to maintaining a high redox (reduction-oxidation) potential and thus maintaining the stability of natural iron oxides in the sediments. That is, nitrate is consumed to yield oxygen before iron oxides, by preference of the active bacteria. Thus nitrates act indirectly to enhance and stabilize the ability of natural iron oxides to bind phosphorus in the sediments. In this manner, nitrate treatment is analogous to hypolimnetic aeration by providing an alternative source of oxygen.

Nitrate treatment is sometimes combined with iron and/or calcium hydroxide treatments to increase effectiveness (Cooke et al., 1993a; 1993b). In Lake Lillesjön, Sweden a harrow was used to distribute the chemicals into the lake bottom. Three chemicals were used: 13 tons $FeCl_3$ (146 g Fe/m^2), 5 tons of slaked lime (180 g Ca/m^2) and 12 tons of $Ca(NO_3)_2$ (141 g N/m^2). All nitrate was denitrified in 1.5 months, but desirable results persisted. A similar treatment was conducted in Lake Trekanten, Sweden, but without the iron and lime (Ripl, 1980). Of the few published accounts, only one (Lake Lillesjön) has shown long-term (ten year) effectiveness (Ripl, 1986).

Nitrate concentrations may increase in the waters where nitrate salts are added. The upper limit for water supplies is 10 mg/L nitrate nitrogen as established by the USEPA. Algal stimulation by nitrate addition is not expected in lakes where phosphorus controls algal growth. In fact, the addition of nitrate may be beneficial even without the stabilizing effect on sediments in some cases, as it would increase the nitrogen to phosphorus ratio, thus benefiting other algal species over the nuisance blue-greens. This is, however, not a widely used technique.

3.5.6 Effectiveness

3.5.6.1 Short-Term

The short-term effectiveness relates to phosphorus precipitation and clarification of the water column. The surface application of the flocculent chemicals (aluminum, iron and calcium) usually has dramatic short-term results. Within hours significant increases in transparency are evident as the floc clears the water of algae and other particulates and concentrations of total phosphorus and reactive phosphorus decline (Jacoby et al., 1994). Where buoyant blue-green algae are abundant, it may take several weeks for these algae to die off, but water clarity improvement will still be noticeable within hours to days. If the lake is stratified, results of injections to the hypolimnion or directly to the sediments may not be apparent until after turnover because the phosphorus in the epilimnetic water is not immediately removed.

3.5.6.2 Long-Term

In cases where P inactivators are added as a flocculation technique, stripping P from the water column, the effectiveness has not been long lasting. This is not surprising, as replacement of the P would be expected with incoming water, with an estimated duration of effects of no more than five times the detention time, based on the standard engineering model of a lake (Metcalf and Eddy, 1972; Weber, 1972). Where detention time is short or treatment is not complete, rapid return to pre-treatment conditions is to be expected. In Martin's Pond for example, water quality data collected by the DWPC indicated that phosphorus levels rebounded to pre-treatment levels within three weeks, while Dug Pond requires annual treatments (L. Lyman, Lycott, pers. comm., 2002a). Beginning in 1989 and in subsequent years, less alum was needed an the water clarity remains at 15-18 feet, thus showing long-term effectiveness of repeated low doses (Lycott Update 2003, Lycott Environmental, Inc.).

Where P inactivation of the sediments is practiced, longevity will depend upon the portion of the total load attributable to internal recycling. Use of alum has provided ten years of improved conditions in shallow lakes and over 15 years of improvement in deep (stratified) lakes (Welch and Cook, 1999) with no follow-up treatment. Cases where alum has failed to provide the desired improvement have universally involved relatively high external loading of P. Iron treatments can remain effective as long as oxygen is present, so use of iron is usually combined with an aeration system. Calcium effectiveness has been less well studied, but results from work in Alberta, Canada suggest that while improvements can last multiple years, the level of improvement is not as large as can be delivered by aluminum or iron/oxygen additions.

Long-term effectiveness relates to how well the phosphorus in the sediment is inactivated and prevented from entering the water column again. This is dose dependent and varies between methods and lakes, but proper assessment of available phosphorus in the sediment, its flux into the overlying water column, and calculation of an appropriate dose of P inactivators should yield long-lived results. For all treatments, if external nutrient loading is relatively high, none of these sediment P inactivation treatments may be very effective.

Another effectiveness issue relates to the availability of hypolimnetic P to the epilimnion over the summer. Cooke et al. (1993b) suggest that strongly stratified lakes may not mix significant phosphorus from the deep bottom waters into the surface during the summer, and thus phosphorus inactivation could have little effect. They conclude that P inactivation is best suited to lakes with an Osgood Index (mean depth in meters/square root of area in km^2) of 6 or less. However, where anoxia is strong enough to produce hydrogen sulfide, a substantial portion of the hypolimnetic phosphorus (typically around 10%, but variable) may diffuse across the thermocline and into the epilimnion. Additionally, where wind is strong, mixing at the boundary of the surficial and deep waters can be a significant source of phosphorus. Consequently, the Osgood Index should not be the sole factor determining applicability.

Wind mixing and redistribution of the floc has been suggested to possibly leave areas of the sediment uncovered (Garrison and Knauer, 1984) or allow inactivated sediment to be buried by new sediment containing available P (Barko et al., 1990). Observations in New England lakes (K. Wagner, ENSR, pers. obs., 1999-2002) indicate that at depths greater than 15 ft, wind processes have minimal effect on alum floc stability. Upon treatment, the floc accumulates on the bottom like a layer of fluffy snow, but gradually condenses and reacts with surficial sediments. In most cases, the floc combines with surficial sediments within a month or two and is not present as a visible layer. Lakes with high sedimentation rates may experience burial of the floc, and new sediment may release phosphorus and reduce the longevity of treatment results. Likewise, wind resuspension in shallow areas may also cause such burial and reduced treatment effects.

3.5.7 Impacts to Non-Target Organisms

3.5.7.1 Short-Term

Aluminum is one of the most common elements on earth, and most organisms are exposed to fairly high levels of aluminum on a regular basis. However, the form of aluminum is especially critical to potential impacts. Reactive aluminum typically undergoes hydrolysis, whereby OH^- radicals are added and a series of tetrahedral compounds are formed. As the molecule grows, it incorporates many other elements and compounds, including the phosphorus that treatments are intended to inactivate. Reactive aluminum has toxic properties, but does not last long in the aquatic environment at pH levels between about 6 and 8 standard pH units. At higher and lower pH values, the potential for toxicity can be significant. Acidic conditions are more common than basic conditions, so aluminum toxicity at low pH is more commonly noted in the literature. Once reacted, however, the resultant aluminum compounds are non-toxic and rather stable.

Short-term effects are therefore more likely than long-term impacts, and involve aluminum toxicity at low or high pH.

Iron and calcium are not known to be toxic at any encountered level. In fact, calcium concentrations above about 3 mg/L are known to reduce the toxicity of aluminum (Baker et al., 1993). The median calcium concentration in Massachusetts is about 5.5 mg/l, but many lakes with calcium less than 3 mg/l are found in the southeast and Cape Cod regions of Massachusetts; the highest values are found in the Berkshires. Silica at levels of 93 µM (5.5 mg/L as SiO_2) has also been suggested to dramatically reduce the toxicity of aluminum (Birchall et al., 1989), although Baker et al. (1990) questioned the results.

No detectable impacts on vertebrates or invertebrates have been observed from calcium treatments (Prepas et al., 1990). Murphy et al. (1988) caution that pH could be elevated to harmful levels if $Ca(OH)_2$ is used as a source of calcium to surface waters. Calcium compounds such as lime are routinely added to domestic water supplies to raise pH (Weber, 1972) and thus no adverse effects are expected from the use of basic calcium compounds in lakes provided that pH remains near the natural level of the receiving waters. However, to get adequate P control with calcium, very high quantities of calcium might have to be added; use of calcium is therefore not appropriate in most Massachusetts lakes, with water bodies in the Berkshires as the only plausible candidates.

Nitrate can displace oxygen attached to hemoglobin molecules at levels >10 mg/L, causing, methemoglobenemia, or blue-baby syndrome. The water quality standard for drinking water has been set at 10 mg/L, although many towns have a more stringent standard for well water at 2-5 mg/L.

In some cases dissolved aluminum concentrations have exceeded the safe level (50-100 µg/L in reactive form), but in most cases detectable fish and invertebrate kills have been avoided. In low alkalinity Kezar Lake, New Hampshire, dissolved aluminum concentrations were as high as 400 µg/L after application of alum and sodium aluminate, but no fish kills were observed. In Lake Morey, Vermont, dissolved aluminum reached concentrations as high as 200 µg/l in the epilimnion where the pH was 8.0 or higher. Despite the high aluminum concentrations, no direct fish mortality was observed. However, the condition of adult yellow perch declined significantly and losses of benthic invertebrates were reported (Smeltzer, 1990). (See section 2.4.4.1). Yet investigations over 14 years since treatment document recovery and marked improvement in the Lake Morey biota, suggesting only temporary impacts (Smeltzer et al., 1999). Laboratory tests indicate very high aluminum levels (80 mg Al/L) can kill invertebrates, possibly by smothering or trapping toxic gases under the heavy floc (Narf, 1990). The eventual incorporation of the floc into the surficial sediments may explain the transient impacts on benthic invertebrates.

A substantial fish kill was reported on May 26, 1995 (Keller, 1995) following aluminum sulfate and sodium aluminate addition to Hamblin Lake in Barnstable, Massachusetts. DFW staff reported an estimate of 16,900 fish killed (Keller, 1995). The fish most impacted appeared to be yellow perch, although rainbow trout, smallmouth bass and brook trout were also killed. The smaller perch were not affected as much as the larger perch and many small perch were observed in schools near the surface after the application. Invertebrates (chironomids and mollusks, but not mayflies) and turtles were also reported. The kill resulted from overbuffering and high pH (values as high as 9.3 SU), leading to aluminum toxicity or possibly pH shock.

A kill similar to that at Hamblin Pond occurred at Lake Pocotopaug in Connecticut in June, 2000, during the early stages of a treatment with a similarly overbuffered mix of alum and aluminate. Fish bioassays revealed behavioral anomalies and up to 30% mortality of juvenile fish after an hour of exposure at pH values as low as 7.5 to 8.0 (ENSR, 2001b). Altering the treatment protocols to set the alum:aluminate ratio at 2:1 (by volume), with application such that total aluminum levels at any point in time were <10 mg/L, resulted in no fish mortality in the lake during completion of the treatment in May 2001. Initial precautions involving the alum:aluminate ratio and application below the thermocline under anoxic conditions resulted in no fish mortality in the 2001 treatment of Ashumet Pond in Mashpee, MA. It now appears possible to perform treatments on low alkalinity lakes without inducing aluminum toxicity.

Other fish kills, much earlier in time, have resulted from lack of buffering of alum treatments. In these cases, the pH dropped to well below 6.0. This has become a rare occurrence, however, as dose adjustments or buffering of treatments in low alkalinity lakes has become standard.

The precipitation of the floc may also carry many other organisms, such as algae and small zooplankton, to the bottom. Changes in the algal community are expected. However, no studies indicate any major shift in zooplankton immediately following treatment. Data for zooplankton in several Maine lakes treated between 1978 and 1986 and monitored before treatment and just after treatment suggest no adverse impacts on zooplankton community composition, density or mean size (Cobbossee Watershed District, unpublished data, 1993).

No adverse impacts on aquatic plants rooted in the sediment have been reported. With increased water clarity, growth of rooted plants at greater depths has been observed. Reduction in the density of plants that depend upon the water column for phosphorus (e.g., duckweed and watermeal) is possible. However, Prepas et al. (1990) reported that some macrophytes were replaced by *Lemna trisulca* after treatment with calcium.

3.5.7.2 Long-Term

Suggested links between aluminum and various diseases have been the subject of debate among toxicologists (Flaten, et al., 1996; Savory, et al., 1996), with no clear consensus regarding the level of risk. There is no active or specific pathway for uptake and retention by man (Duffield and Williams, 1989). Normal ingestion rates for humans are expected to range from 1 to 10 mg per day (Sherlock, 1989), and some aluminum salts are used in commonly available stomach antacids, but nearly all ingested aluminum is biologically unavailable (Duffield and Williams, 1989). If small amounts do enter the blood stream, they are rapidly excreted by normal renal mechanisms (Duffield and Williams, 1989). Aluminum may be a health problem in people with kidney dysfunction (Stewart, 1989), as a function of its coagulant properties while in a reactive form. Aluminum has been associated with a 1.5x increase in Alzheimer's disease in areas where aluminum exceeded 0.11 mg/L in the public water supply (Martyn, 1989), but this is a correlation, not a cause and effect relationship. Note that alum and aluminum salts are commonly used for coagulation and flocculation processes to clarify water supplies and in wastewater treatment (Weber, 1972).

Although some short-term effects have been noted, there do not seem to be any significant long-term impacts on benthic invertebrates (Smeltzer et al., 1999; K. Wagner, ENSR, pers. obs., 1999-2002). As an exception, short-term toxicity testing showed no effects on midge larvae, while chronic tests over 55 days showed 37% mortality at a 10 mg Al/L dose compared to 5.4% in the control. Yet the mechanism of mortality was unknown (Lamb and Bailey, 1983), and this is a high dose for that duration of study. Despite the potential toxicity, and considering the high alum application rates, few adverse effects are reported (e.g. Jacoby et al., 1994). Benthic invertebrate density may actually increase within a season (Narf, 1990; Smeltzer et al., 1999). In one case of long-term alum treatment upstream of Lake Rockwell, Ohio, alum caused reductions in invertebrates either by toxicity or downstream drift of the organisms in the river (Barbiero et al., 1988), but in this case the treatments were repetitive and frequent.

Bioaccumulation of aluminum has not been reported. No impacts on trout were observed over one month (Lamb and Bailey, 1983). A long-term study following treatment of Kezar Lake, New Hampshire found some changes in zooplankton as cladoceran crustacea declined (Connor and Martin, 1989). Such changes may be naturally expected if algal food supplies decline and visual predation increases following treatment. Reducing algal production might be expected to reduce fish production as well. On the other hand, increased transparency may allow macrophytes to increase and extend their depth distribution into deeper waters as sunlight penetration increases.

3.5.8 Impacts to Water Quality

3.5.8.1 Short-Term

The chemistry of aluminum in treated water has been reviewed by Driscoll and Letterman (1988). Alum is acidic and can drive pH down below 6 SU, causing dissolved aluminum concentrations remain high for a longer period of time than at more moderate pH. Sodium aluminate is basic and drives pH upward beyond pH 9 where dissolved aluminum concentrations may remain high for a longer period of time than at more moderate pH. All other effects of aluminum on water chemistry are related to the removal of a variety of contaminants from solution by coagulation and precipitation. Aluminum is used extensively in the water treatment industry for its rapid coagulant benefits to water quality.

As long as oxygen levels are suitably high, iron behaves much like aluminum sulfate in terms of its effects on pH and water quality contaminant levels. Calcium compounds raise the pH, but also are expected to remove many contaminants.

3.5.8.2 Long-Term

No direct adverse long-term impacts on water quality are expected and none have been reported for any of these treatments. Indirect changes are expected to be beneficial; the intended long-term change is a reduction in available phosphorus, which in turn should improve water quality by reducing algal production and associated fluctuations in pH, oxygen and solids in the water column.

3.5.9 Applicability to Saltwater Ponds

While these treatments may work in saltwater ponds, little information is available on any such experience. Saltwater ponds may be highly stratified with saltwater below and freshwater above. In such cases, mixing estimates may be required to calculate the potential for mixing of phosphorus to the surface waters and to evaluate the applicability of such treatments. Potential effects of flocs on shellfish in saltwater ponds would be a primary concern.

3.5.10 Implementation Guidance

3.5.10.1 Key Data Requirements

This nutrient control method requires an accurate nutrient budget that includes both a measured mass balance and a land-use source analysis, and it should include a detailed analysis of internal sources of phosphorus (Section 1). If the nutrient budget shows that the major source of phosphorus is from the sediments, then these types of nutrient controls may be effective. Even in lakes where there are large external sources, these treatments (especially alum and calcium) will clarify the water. However, the effectiveness may not last more than a year or two (possibly as short as a few weeks depending on detention time) if the external sources are not controlled as well. Alum, iron and calcium treatments require recent information on pH and alkalinity at all depths to properly predict potential changes in pH and to minimize impacts. Knowledge of lake oxygen regime and biotic components is helpful in planning treatments. An accurate depth map of the lake is required to properly evaluate dosing. In addition to jar tests to establish doses and ratios of chemicals, toxicity tests with a sensitive fish species such as fathead minnow may be desirable to ensure the safety of the treatment. Estimates of effectiveness should be made for lake recovery in terms of total phosphorus levels and Secchi disk transparency. For deep lakes, hypolimnetic dissolved P concentration should decrease dramatically and should be checked.

3.5.10.2 Factors that Favor this Approach

The following considerations are indicative of appropriate application of phosphorus inactivation for reductions in nutrient concentrations and control of algae in lakes:

1. A substantial portion of the P load is associated with sediment sources within the lake.
2. Studies have demonstrated the impact of internal loading on the lake.
3. External P load has been controlled to the maximum practical extent or is documented to be small; historic loading may have been much greater than current loading.
4. Inactivation of phosphorus in the water column is expected to provide interim relief from algal blooms and turbidity while a prolonged watershed management program is conducted to reduce external loading.
5. The lake is well buffered or buffering can be augmented to prevent major changes in pH during treatment.
6. Assays indicate no toxic effects during simulated treatment.
7. Where iron is to be used as an inactivator, oxygen is adequate at the bottom to maintain iron-phosphorus bonds.
8. Where calcium is to be used as an inactivator, normal background pH is high enough to maintain calcium-phosphorus bonds.

Where nitrate is to be used to alter redox potential and limit P release, nitrate can be effectively injected into the sediment without major release to the water column.

3.5.10.3 Performance Guidelines

Planning and Implementation

Treatments for phosphorus inactivation need to be carefully planned and executed to achieve the desired goal without undue impact to non-target organisms. In most cases the primary goal will be long-term reduction in internal P recycling, but may be short-term reduction of P in the water column until watershed management can reduce P loading. If short-term reduction is to be a repetitive process for an indefinite number of years, further consideration of impacts may be warranted and cost comparison with watershed management over a longer period (10-20 years) is encouraged.

Access and a staging area for loading of chemicals is needed for efficient treatment, and project size will determine many other needs. Broadcasting powdered inactivators is possible but generally restricted to smaller applications (<10 acres). Application of liquids usually involves loading one or more tanks on a boat, barge, or modified harvester with frequent refills. Targeted treatment areas should be clearly laid out. Geographic positioning systems are now commonly used, but demarcation with buoys is still a desirable back-up plan. Equipment for injecting the chemicals well below the water surface may be needed.

Alum, iron and calcium compounds can all be injected into the hypolimnion of deeper lakes to minimize impacts to the surface waters. Alternatively, the lake can be treated in sections over time to maintain refuge areas for fish, or the lake could be treated multiple times with lower doses. Buffers can be added to maintain appropriate pH in low alkalinity lakes treated with alum or iron, and for treatments in which aluminum sulfate is buffered by sodium aluminate, a 2:1 ratio of alum to aluminate by volume is recommended. Calcium or silica additions may be considered to reduce aluminum toxicity during alum treatments. Calcium additions for phosphorus inactivation may be difficult to perform effectively in many Massachusetts lakes, given low pH in all but the lakes of the Berkshires. Iron treatments may require aeration to maintain oxic conditions. Nitrate treatments are generally injected directly into the sediments and thus should not impact surface waters in any major way.

Monitoring and Maintenance

Chemical samples for total phosphorus, dissolved cations (aluminum, iron or calcium, depending upon the treatment), alkalinity and pH should be collected and analyzed before, during and after treatment at several depths (typically 10 ft intervals). Nitrate levels should be monitored in the case of a nitrate treatment. Shifts in alkalinity and pH are most important to track during treatment, the former providing a warning of possible impacts to the latter. Dissolved oxygen might also be monitored during treatment and as part of a long-term monitoring program; temporary decreases in deeper water dissolved oxygen may occur, followed by longer term increases. Long-term monitoring of water clarity is the simplest measure of treatment effectiveness and longevity.

Pre- and post-treatment biological sampling should include identification and enumeration of algae and zooplankton and a visual survey for any large impacts (e.g., macroinvertebrate or fish kills). Where sensitive populations reside in the treatment area and have little opportunity to vacate the area during treatment, some pre- and post-treatment monitoring of those populations may be warranted.

Generally little maintenance is required for these treatments. Ideally, treatments for inactivation of sediment P are one-time efforts for any lake within the lifetime of the applicants. For maintenance water column treatments, re-application is the only maintenance activity.

Mitigation

Once a treatment is applied, there is little opportunity for mitigation. Performing a treatment over time, with sections of lake treated on separate days with a period of evaluation in between, can allow adjustment of cancellation of treatment where adverse impacts are detected. This is a reasonable approach on larger projects, but may not be effective for small lake treatments and adds considerably to the cost.

3.5.11 Regulations

3.5.11.1 Applicable Statutes

In addition to the standard check for site restrictions or endangered species (see Appendix II.), a Notice of Intent must be sent to the Conservation Commission with a copy to the Department of Environmental Protection Regional Office. If the proposed project occurs within an Estimated Habitat of Rare Wildlife in the most recent version of the Natural Heritage Atlas, a copy of the Notice of Intent must be submitted to the Natural Heritage and Endangered Species Program (NHESP) within the MDFG for review (Appendix II). If the proposed project occurs within a Priority Habitat of Rare Species in the most recent version of the Natural Heritage Atlas, the project proponent must submit project plans to the NHESP for an impact determination. An Order of Conditions must be obtained prior to work. Check threshold requirements for MEPA review (Appendix II). A License to Apply Chemicals is required, but the applicator is not required to be licensed by the Massachusetts Department of Agricultural Resources. A Chapter 91 Permit is not required for phosphorus inactivation treatments (Appendix II) and the Corps of Engineers does not regard nutrient inactivation as a filling of wetland resources, so no Section 404 permit is required. A Section 401 permit is sometimes required from the MDEP, depending upon funding source and the details of other permits.

3.5.11.2 Impacts Specific to Wetlands Protection Act

The following overall impact classification is offered as a generalization of impacts, with clarifying notes and caveats as warranted:

1. Protection of public and private water supply – Benefit (water quality improvement).
2. Protection of groundwater supply – Neutral (no significant interaction).
3. Flood control - Neutral (no significant interaction).
4. Storm damage prevention – Neutral (no significant interaction).
5. Prevention of pollution - Benefit (water quality enhancement).
6. Protection of land containing shellfish - Possible benefit through water quality enhancement in the lake and possible detriment by direct toxicity unless treatment is properly buffered.
7. Protection of fisheries - Possible benefit through water quality enhancement in the lake and possible detriment by direct toxicity unless treatment is properly buffered, plus possible detriment through reduced fertility.

8. Protection of wildlife habitat – Benefit (water quality enhancement), but possible detriment through reduced fertility.

The most serious impact is the possibility for fish or invertebrate kills following treatment in low alkalinity lakes. This can be avoided with proper planning and implementation. Minimal adverse impacts are expected to either surface or groundwater supplies. Aluminum, iron and calcium are commonly added in water and wastewater treatment facilities with no significant adverse impacts (and generally a marked improvement in water quality). However, nitrate could adversely impact water supplies if levels in the water approach the 10 mg/L limit, and could disrupt lake ecology at levels as low as 0.5 to 1.0 mg/L. Yet nitrate treatment acts directly on sediment and is not expected to raise nitrate levels in the water column.

3.5.12 Costs

Costs vary by the amount (dose) applied, the total area treated, and by the precautions necessary to avoid unintended impacts.

3.5.12.1 Aluminum

Aluminum treatment costs typically range from $500-$1,00/acre, with the areal cost decreasing for larger treatments, unbuffered treatments, and lesser monitoring requirements (Wagner, 2001). Total cost was $47,000 for treatment of 86 acres ($546/acre) of Hamblin Pond in Barnstable, MA. This was a fairly typical inactivation effort, but one that resulted in a fish kill due to improper buffering, application near the surface, and inadequate monitoring. Treatment of Ashumet Pond cost $337,000 for 28 acres ($12,000/acre), owing to extensive pre-treatment planning, permitting, and testing, treatment at 35 ft of water depth, an extreme amount of monitoring during treatment, and oversight by three consulting firms. This level of treatment cost is simply not sustainable by most applicants. Costs for treatment of Dug Pond are about $300 to $400/acre, but this is a low-dose, annual maintenance treatment that does not provide long-term P control. Costs for treating Mountain and Cranberry Lakes in NJ with lime and alum on a maintenance basis range from $200 to $1,000/acre on annualized scale, depending on the frequency of treatments (every other year to 2/yr). The treatment of 177 acres of Lake Pocotopaug in CT cost about $220,000 (about $1,250/acre), including a thorough investigation of the initial fish kill and extensive monitoring.

3.5.12.2 Iron

Costs for iron treatments are similar to those for alum treatment; the chemical is less expensive to purchase but higher doses are recommended (100 g Fe/m^2) (Cooke et al., 1993a). However, iron is best applied in conjunction with aeration systems, so total project cost is likely to be substantially higher.

3.5.12.3 Calcium

Calcium costs are slightly less expensive than alum, especially in hard water lakes where this technique is most likely to be applied. The cost is estimated at $10 per metric ton for $CaCO_3$ and $100 per metric ton of $Ca(OH)_2$ from work done in Alberta, Canada. Due to the nature of calcite solubility, more of the former was required to achieve the desired results. Thus, the cost of materials is about $200/acre. Labor has been a non-commercial cost in most calcium treatments, conducted by University of Alberta researchers.

3.5.12.4 Nitrate

Nitrate application to sediments is an expensive treatment. At White Lough the costs were estimated to be 80% higher than alum treatment, even at nitrate doses 5 times lower than that applied at Lake Lillesjön (Foy, 1986). This is largely due to the high cost to inject the chemical into the sediment.

3.5.13 Future Research Needs

Evaluation of the right level of precaution and monitoring is needed to make inactivation both safe and affordable. Application of all discussed precautions will tend to be overprotective and greatly adds to treatment cost. Further testing is needed on the use of calcium and silica to ameliorate possible impacts of alum treatments, if this approach is to be developed.

3.5.14 Summary

Phosphorus inactivation offers one of the most effective long-term management options for eutrophic lakes suffering from algal blooms if the source of the phosphorus is the lake sediments. In cases where large inputs of phosphorus are coming from watershed or point sources, these should be addressed first. However, interim inactivation of phosphorus in the water column on a seasonal basis may be an appropriate maintenance technique while prolonged watershed management actions are underway. Of the four types of treatments, alum (with or without buffers) has the most proven record of effectiveness. Iron, calcium and nitrate treatments may be applicable under certain circumstances, but suffer from limitations that affect the level of success and longevity of results. In lakes where sediments are a major source of phosphorus, a single large treatment (inactivation of sediment phosphorus) can provide rapid and lasting relief from elevated phosphorus and algae levels. Although alum treatments can have adverse impacts, including fish or invertebrate kills, the method can be used if proper precautions are taken. Extra precautions are needed in low alkalinity lakes (< 20 mg/L).

3.6 ARTIFICIAL CIRCULATION AND AERATION

3.6.1 Overview

Whole lake circulation and hypolimnetic aeration are two related techniques for management of algae that tend to affect nutrient levels. The central process is the introduction of more oxygen, intended to limit internal recycling of phosphorus, thereby controlling algae. Other potentially important processes may be at work here as well, however. Circulation strategies minimize stratification, while hypolimnetic aeration maintains stratification (Figure 3-1).

Whole lake artificial circulation is also referred to as destratification or whole lake aeration. Circulation affects mixing and the uniformity of lake conditions. Thermal stratification and features of lake morphometry such as coves create stagnant zones that may be subject to loss of oxygen, accumulation of sediment, or algal blooms. Artificial circulation minimizes stagnation and can eliminate thermal stratification or prevent its formation. Movement of air or water is normally used to create the desired circulation pattern in shallow (<20 ft) lakes, and this has been accomplished with surface aerators, bottom diffusers, and water pumps. Algae may simply be mixed more evenly in the available volume of water in many cases, but turbulence, changing light regime and altered water chemistry can cause shifts in algal types.

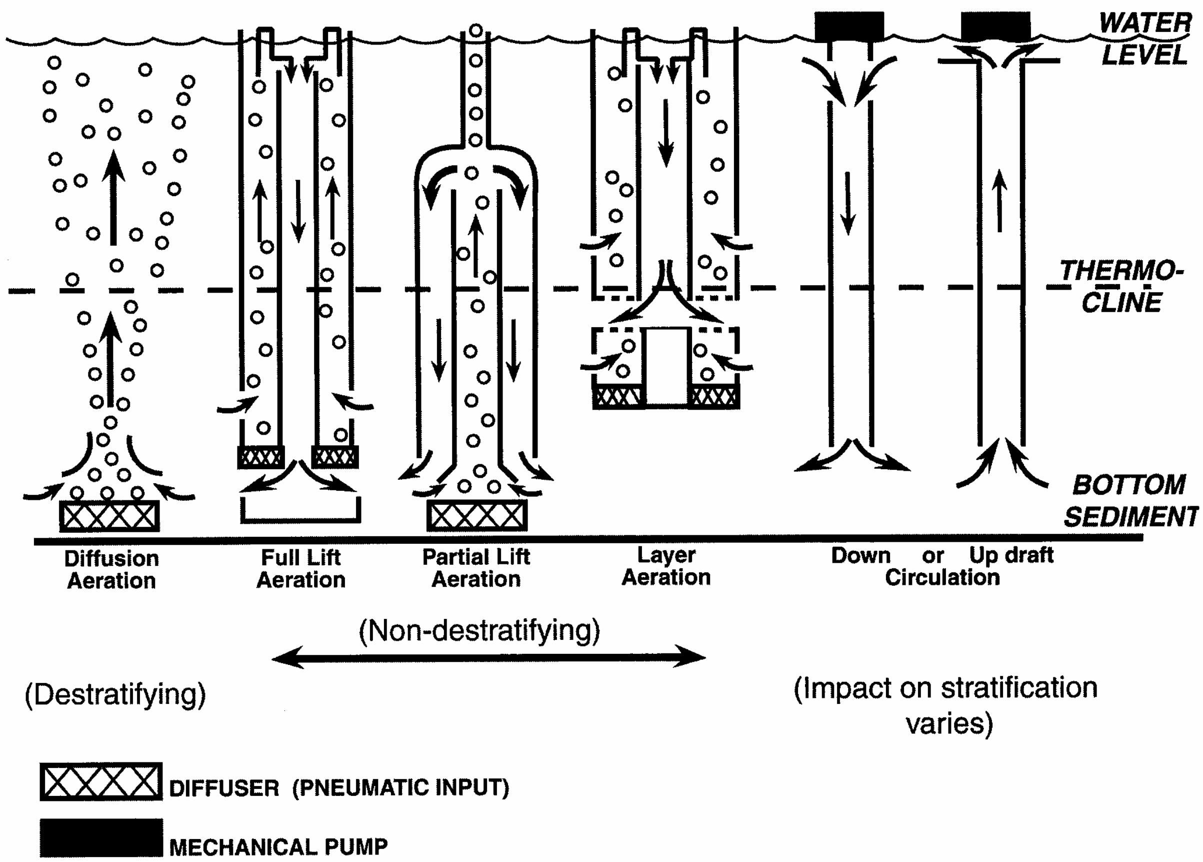

Figure 3-1 Methods of artificial circulation and aeration (from Wagner, 2001)

Stratification is broken or prevented in deeper lakes through the injection of compressed air into lake water from a diffuser at the lake bottom. The rising column of bubbles, if sufficiently powered, will produce lakewide mixing at a rate that eliminates temperature differences between top and bottom waters. The use of air as the mixing force also provides some oxygenation of the water, but the efficiency and magnitude of this transfer are generally low. In some instances, wind driven pumps have been used to move water. For air mixed systems, the general rule is that an air flow rate of 1.3 cubic feet per minute per acre of lake (9.2 m^3/min/km^2) will be needed to maintain a mixed system (Lorenzen and Fast, 1977). However, there are many factors that could require different site specific air flow rates, and undersizing of systems is the greatest contributor to failure for this technique.

Algal blooms are sometimes controlled by destratification through one or more of the following processes:

- Introduction of dissolved oxygen to the lake bottom may inhibit phosphorus release from sediments, curtailing this internal nutrient source.
- In light-limited algal communities, mixing to the lake's bottom will increase the time a cell spends in darkness, leading to reduced photosynthesis and productivity.
- Rapid circulation and contact of water with the atmosphere, as well as the introduction of carbon dioxide-rich bottom water during the initial period of mixing, can increase the carbon

dioxide content of water and lower pH, leading to a shift from blue-green algae to less noxious green algae.
- Turbulence can neutralize the advantageous buoyancy mechanisms of blue-green algae and cause a shift in algal composition to less objectionable forms such as diatoms.
- When zooplankton that consume algae are mixed throughout the water column, they are less vulnerable to visually feeding fish. If more zooplankters survive, their consumption of algal cells may also increase.

Some of the early applications of artificial circulation were to prevent winterkills of fish in eutrophic lakes that become anoxic during the winter. On a smaller scale, artificial circulation can be used to prevent ice formation around docks or other structures. The technique is also used to maintain acceptable water quality in drinking reservoirs as the oxic conditions created by the circulation reduce concentrations of nuisance substances such as hydrogen sulfide, ammonia, iron and manganese. For these types of problems artificial circulation has been very successful.

Hypolimnetic aeration typically uses an air compressor as described for whole lake circulation above, but in this case the upward plume is controlled to avoid mixing with the epilimnetic waters, and thus thermal stratification of the lake is maintained. The maintenance of stratification is often desirable as it maintains coldwater fish habitat and reduces transport of nutrients from the hypolimnion into the epilimnion where they may stimulate further algal blooms.

Aeration puts air into the aquatic system, increasing oxygen concentration by transfer from gas to liquid and generating a controlled mixing force. The oxygen transfer function is used to prevent hypolimnetic anoxia. By keeping the hypolimnion from becoming anoxic during stratification, aeration should minimize the release of phosphorus, iron, manganese and sulfides from deep bottom sediments and decrease the build-up of undecomposed organic matter and oxygen-demanding compounds (e.g., ammonium). Hypolimnetic aeration can also increase the volume of water suitable for habitation by zooplankton and fish, especially coldwater forms. Pure oxygen can be used in place of air to maximize oxygen transfer at an increased cost.

A full lift hypolimnetic aeration approach moves hypolimnetic water to the surface, aerates it, and replaces it in the hypolimnion. Bringing the water to the surface can be accomplished with electric or wind-powered pumps, but is most often driven by pneumatic force (compressed air). Return flow to the hypolimnion is generally directed through a pipe to maintain separation of the newly aerated waters from the surrounding epilimnion. To provide adequate aeration, the hypolimnetic volume should be pumped and oxygenated at least once every 60 days.

Another hypolimnetic aeration system is the partial lift system, in which air is pumped into a submerged chamber in which exchange of oxygen is made with the deeper waters. The newly oxygenated waters are released back into the hypolimnion without destratification. A shoreline site for a housed compressor is needed, but the aeration unit itself is submerged and does not interfere with lake use or aesthetics.

An alternative approach involves a process called layer aeration (Kortmann et al. 1994). Water can be oxygenated by full or partial lift technology, but by combining water from different (but carefully chosen) temperature (and therefore density) regimes, stable oxygenated layers can be formed anywhere from the upper metalimnetic boundary down to the bottom of the lake. Each layer acts as a barrier to the passage of phosphorus, reduced metals and related contaminants from the layer below. Each layer is stable as a consequence of thermally mediated differences in density. The whole hypolimnion may be aerated, or any part thereof, to whatever oxygen level is deemed appropriate for the designated use.

The mechanism of phosphorus control exercised through hypolimnetic aeration is the maintenance of high oxygen and limitation of phosphorus release from sediments. Out of the processes listed for artificial circulation, the only other applicable mechanism for hypolimnetic aeration is provision of a zooplankton refuge, potentially increasing grazing potential. To successfully aerate a hypolimnion, the continuous oxygen demand of the sediments must be met, and experience dictates that the oxygen input needs to be about twice the measured oxygen demand (Cooke et al., 1993a). This demand may be reduced over time under aeration, but is unlikely to be eliminated.

It is also essential that an adequate supply of phosphorus binder, usually iron or calcium, be available to combine with phosphorus under oxic conditions. Sediments are likely to be anoxic below the surface, even with a well-oxygenated water column above, so some release is to be expected unless phosphorus binders are sufficient to immediately combine with phosphorus at the anoxic-oxic interface. In many cases iron will be solubilized with the phosphorus, and can recombine with it upon oxygenation. However, where sulfate reduction is active, iron may be scavenged by sulfides and be unavailable for binding phosphorus (Gachter and Muller, 2003). In such cases, additional phosphorus binders may have to be added for aeration to have maximum effectiveness on phosphorus inactivation.

3.6.2 Effectiveness

The success of circulation or aeration in controlling algae is largely linked to reducing available phosphorus, which in turn depends on detailed aspects of system respiration and the chemical content of the water and bottom sediment. Enough oxygen must be added to meet the oxygen demand, and there must be an adequate supply of phosphorus binders present. If phosphorus binding agents are naturally insufficient, results can be improved by adding reactive aluminum or iron compounds to the process. Circulation may provide additional benefits through altered pH or other water chemistry in surface waters, or by subjecting algae to variable light regime and physical stresses associated with mixing. Both circulation and hypolimnetic aeration can foster more desirable zooplankton communities, increasing grazing on algae.

3.6.2.1 Short-Term

Short-term effectiveness may be achieved if oxygen levels near the bottom rise quickly and adequate phosphorus binders are present. Even then, a month or more of lag time might be expected for existing algae to suffer nutrient limitation or other stresses that reduce abundance. The control of phosphorus in surface waters may not be effective until the following year for hypolimnetic aeration.

The use of artificial circulation to control algal blooms has had varied results. A review by Pastorok et al. (1982, as cited in Cooke et al., 1993a) of many whole lake artificial circulation treatments found that in more than half of the cases conditions became worse. Total phosphorus increased or did not change in 65% of the cases, Secchi disk depth worsened in 53% of the cases and phytoplankton decreased in less than half of the cases. The technique is sometimes effective at shifting phytoplankton composition from blue-greens to green algae or diatoms. Despite the lack of consistent evidence of lake improvement, aeration is a popular technique for owners of small ponds, where it is claimed to reduce algae in addition to providing additional oxygen for fish populations (Matson, 1994). Destratification has been a very successful technique for drinking water reservoirs, as evidenced by Fresh Pond in Cambridge, but there it is the build-up of manganese under anoxic conditions that is being counteracted in this generally low-nutrient system.

Several reasons for failure of whole lake circulation to achieve consistent algal reduction have been suggested, mostly related to improper design or placement of pneumatic mixing systems. In Silver Lake, Ohio, Brosnan and Cooke (1987) found that artificial circulation failed to improve the eutrophic conditions. Insufficient mixing caused by an underpowered air compressor was suggested as a possible reason for the failure; the airflow was only three percent lower than the calculated target rate of 3.5 m^3/min/ km^2, but the target rate itself may have been low. High mixing rates are reported to be more effective than low mixing rates, with the recommended air-flow rate of 9.2 m^3/min/km^2 (Lorenzen and Fast, 1977) marking the boundary between high and low rates. The higher mixing rates prevent microstratification that can allow algae to remain in the photic zone and result in an algal bloom (Brosnan and Cooke, 1987). Another reason suggested to explain the increases in total phosphorus has been resuspension of nutrient-rich sediments caused by improper placement of the diffusers directly on the sediment surface (Brosnan and Cooke, 1987).

Hypolimnetic aeration has had generally positive results, but effectiveness has been variable. Cooke et al. (1993) review a number of examples and note that available phosphorus tends to decline by one to two thirds during aeration, but often rises quickly to pre-aeration levels when treatment ceases. Aeration promotes binding activity and has been most effective when phosphorus binders have been added. Sedimentation of previously available phosphorus in a Canadian lake increased by almost an order of magnitude after aeration with the addition of iron to a Fe:P ratio of 10:1 (McQueen et al., 1986a), and the combination of iron and oxygen was similarly successful in a Minnesota Reservoir (Walker et al., 1989). Aluminum can minimize phosphorus availability even in the absence of oxygen. The process of nutrient inactivation is covered separately in this document, but the synergy of these techniques is notable, and aeration depends to a large degree on the availability of phosphorus binders to reduce phosphorus levels.

Hypolimnetic aeration has been reported to be reasonably successful (Kishbaugh et al., 1990; Wagner, 2001), but in many cases little improvement has been reported. Multiple factors may be responsible, one of which is continued metalimnetic anoxia, where organic particles accumulating near the thermocline create an anoxic layer above the aerated hypolimnion. A successful example of an increase in transparency and reduction in blue-green algae in a Connecticut lake is described in Kortmann et al. (1994) who used layer aeration within the thermocline of a eutrophic water supply lake. The authors suggest that layer aeration (where the oxygenated water is used to create a stable layer instead of aerating the entire hypolimnion) can eliminate the problem of metalimnetic anoxia that allows rapid phosphorus recycle and can act as a barrier to fish migration.

Any aeration system can make a marked improvement in lake conditions, but it should be noted that practical experience has demonstrated that effects are not uniform or consistent within and among aquatic systems. Zones of minimal interaction will often occur, possibly resulting in localized anoxia and possible phosphorus release. Partial lift hypolimnetic aeration systems may allow a band of anoxic water to persist near the top of the metalimnion, allowing nutrient cycling and supply to the epilimnion and discouraging vertical migration by fish and zooplankton. Phosphorus binders must be available for aeration to facilitate phosphorus inactivation. Uniformity of results should be achievable with careful design and operation, but probably with increased cost.

3.6.2.2 Long-Term

Since aeration is an active treatment, the pumps must be kept running year after year, at least during the summer months, but it seems plausible that effectiveness can be maintained over many years with this method. Certainly the Fresh Pond destratification system in Cambridge has yielded positive results over a period approaching a decade. Notch Reservoir in North Adams has also experienced improvement over about a decade with a hypolimnetic aeration system. Kortmann et al. (1994) describe a successful long-term treatment of Lake Shenipsit, Connecticut with a layer aeration method. In this case, aeration was conducted for several years between 4.7 and 10.7 meters in a lake with a maximum depth of 20.7 meters. Adequate aeration of the metalimnion in this 212 ha lake was achieved with compressor systems totaling 60 HP that delivered 240 CFM or 6.8 $m^3/min/km^2$. Total phosphorus was reduced marginally while blue-green algae decreased and the algal community shifted to green algae and diatoms. The lake experienced a large increase in transparency after 2 years of layer aeration. The increase was associated with an increase in zooplankton, particularly *Daphnia*, that were assumed to be grazing on the algae (Kortmann et al., 1994) and may have used the newly oxygenated zone as a daytime refuge from fish predation.

3.6.3 Impacts to Non-Target Organisms

3.6.3.1 Short-term

There are very few negative impacts expected from hypolimnetic aeration but several potentially adverse impacts from circulation. In general, however, these techniques have limited potential to cause any harm if properly designed. Since oxygen levels are increased in previously anoxic area, many organisms that require oxygen such as fish, aquatic insects and zooplankton are expected to increase for both whole lake circulation and hypolimnetic aeration (Pastorok et al., 1980). Ashley (1983) noted an increase in some zooplankton species following hypolimnetic aeration, despite little effect on algae.

The greatest risk from artificial circulation involves transport of nutrients and other substances from the bottom to the top of a lake. If the bottom waters are rich in nutrients and the epilimnion low in nutrients, whole lake circulation may transport nutrients (and possibly silt) to the surface and stimulate unwanted algal blooms and reduce transparency in the surface waters (Cooke et al., 1993a; Brosnan and Cooke, 1987). Algal blooms could lower epilimnetic carbon dioxide, raise pH and possibly lead to blue-green dominance as suggested by Cooke et al. (1993). Changes in zooplankton and algal communities could have an effect on the fish populations in the higher trophic levels.

Another risk from artificial circulation involves altered thermal regime. For whole lake circulation, the temperature increase in the bottom waters may be considered an adverse effect since it may eliminate cold water fishery habitat from the lake if the water becomes too warm. However, it is expected that this method would be used in eutrophic lakes with anoxic hypolimnia, where no significant cold water fishery was present due to the lack of oxygen.

The need to continue to aerate or circulate is an important consideration. While cessation may not result in worse conditions than encountered before treatment, adjustment of system biota to the new oxygen or thermal regime could be a problem. In one case a fish kill was reported in a water supply reservoir (Mt. Williams Reservoir, North Adams, MA) during a period of high turbidity when a destratifying aerator was turned off in 1993 (DFW, unpublished data, 1993).

3.6.3.2 Long-Term

Long-term impacts to biota such as zooplankton and fish may occur following any changes in algal abundance or species composition. Cold water fisheries may be harmed if the cold thermal refuge is eliminated by mixing. Oxygen or nitrogen supersaturation could theoretically become a problem for fish in deep waters during aeration due to gas bubble disease, but formation of the right size bubbles from aeration is not expected (Cripe and Phipps, 1999). Gas bubble disease is most often a function of creation and entrapment of very fine air bubbles associated with hydropower facilities; aeration systems have not been observed to produce bubbles small enough to induce this disease. Cooke et al. (1993) suggest that nitrogen supersaturation represents a greater risk than oxygen levels, but that no gas bubble disease has been detected in lakes with hypolimnetic aeration. Although gas bubble disease is known to occur near deep groundwater springs and below large hydropower dams (Marking, 1987), no cases of gas bubble disease have been reported in the many lakes and reservoirs where aeration is used.

Direct impacts on humans are mostly safety and noise related. If aerators are operated during the winter months (to prevent fish kills or protect structures) then the aeration sites should be clearly marked as thin ice or no ice areas to minimize the hazard to winter lake users. Deaths from drowning have been known to occur under such circumstances (NRC, 1992). Ellis and Stefan (1994) have proposed and tested a method to preserve ice cover during aeration operations that would reduce this hazard. Noise from compressors or pumps can be an issue for nearby residents, and usually is mitigated by placing these machines in buildings that suppress noise.

3.6.4 Impacts to Water Quality

3.6.4.1 Short-Term

In most cases water quality improves as elevated oxygen levels reduce the concentrations of phosphorus, hydrogen sulfide, iron and manganese that are commonly found in anoxic waters. Algal production should decline as a function of reduced phosphorus availability, but this has not always been the case, especially with circulation systems that mix the lake to varying degrees and may actually increase nutrient availability. If installed too close to the sediments, diffusers may resuspend sediments causing increases in turbidity and an increase in total phosphorus. Even when properly installed, turbidity may increase somewhat.

3.6.4.2 Long-Term

Long-term impacts on water quality are essentially the same as the short-term impacts, with an intended improvement in water quality as described above (Verner, 1984; Boehmke, 1984). It is unlikely that a circulation or aeration system would be operated for more than a few years if such a water quality improvement was not observed.

3.6.5 Applicability to Saltwater Ponds

This technique could be applied to saltwater ponds, although no such applications have been reported. In some saltwater ponds, very strong density gradients can occur if relatively fresh water is present over a saltwater hypolimnion, and additional calculations would be required to determine how much energy would be required to circulate such a system. In addition, a whole lake circulation may have adverse impacts associated with the osmotic shock that would occur when freshwater biota become mixed into saltwater and vice versa. Suspended sediments may interfere with filter feeding of shellfish if care is not taken during installation. Overall, however, there is no reason to believe that the addition of oxygen to saltwater ponds would not be beneficial.

3.6.6 Implementation Guidance

3.6.6.1 Key Data Requirements

Ideally, data related to each of the five possible control mechanisms (oxygenation/P inactivation, light limitation, pH/carbon source adjustment, buoyancy disruption, and enhanced grazing) should be analyzed and evaluated in terms of potential algal control. Specifically,

1. Is there anaerobic release of phosphorus that can be mitigated by oxygenation of deep waters?
2. Is the supply of phosphorus binders adequate to inactivate most phosphorus upon oxygenation?
3. Is the mixing zone deep enough to promote light limitation of algae?
4. Is there a large amount of carbon dioxide in the bottom waters that could be mixed to the surface to favor the growth of algae other than blue-greens?
5. Is mixing predicted to counteract the buoyancy advantage of blue-greens over other algae?
6. Will a dark, oxygenated refuge be created for zooplankton?

Of the five mechanisms, oxygenation to prevent sediment release of phosphorus is the best documented and should be the focus of most treatments of this type. If the nutrient budget does not indicate a large source of phosphorus-rich, anaerobic water in the hypolimnion, these methods are not as likely to be successful. Data requirements for this type of nutrient control therefore include an accurate nutrient budget with a detailed analysis of internal sources of phosphorus (Section 1) and availability of potential phosphorus binders.

The most critical information for designing an aeration system is the oxygen demand that must be met by the system. Oxygen demand is normally calculated from actual data for the lake. For stratified lakes, the hypolimnetic oxygen demand (HOD, often a function of sediment oxygen demand, or SOD) can be calculated as the difference in oxygen levels at the time stratification formed and one or more points in time later during stratification. However, measurements obtained when the oxygen levels are <2 mg/L are deceiving, as oxygen consumption is not linear and will decline markedly as oxygen supply declines. Oligotrophic lakes typically have oxygen demands <250 $mg/m^2 \cdot day$, while eutrophic lake values are >550 $mg/m^2 \cdot day$ (Hutchinson 1957). Hutchinson suggests that 1400 $mg/m^2 \cdot day$ is the upper boundary for eutrophic lakes; values of 2000 to 4000 $mg/m^2 \cdot day$ have been measured in hypereutrophic lakes (K. Wagner, ENSR, pers. obs., 1996-2000). There are a number of other factors complicating the assessment of oxygen demand; calculations and related interpretation for design purposes are best performed by experienced professionals.

3.6.6.2 Factors that Favor this Approach

The following considerations are indicative of appropriate application of artificial circulation and hypolimnetic aeration for reductions in nutrient concentrations and control of algae in lakes:

1. A substantial portion of the P load is associated with anoxic sediment sources within the lake.
2. Studies have demonstrated the impact of internal loading on the lake.
3. External P load has been controlled to the maximum practical extent or is documented to be small; historic loading may have been much greater than current loading.
4. Hypolimnetic or sediment oxygen demand is high (>500 $mg/m^2/day$).
5. In addition to phosphorus management, control of other reduced compounds such as hydrogen sulfide, ammonia, manganese and iron, is desired.
6. Adequate phosphorus inactivators are present for reaction upon addition of oxygen.
7. Shoreline space for a compressor or pump is available where access is sufficient and noise impacts will be small.
8. Power is available to run all machinery.
9. The lake is bowl shaped, or at least not highly irregular in bathymetry (few separate basins and isolated coves).
10. Long-term application of the technique is accepted.
11. For artificial circulation, coldwater fishery habitat is limited or not a concern.
12. For hypolimnetic aeration, coldwater fishery habitat is abundant or an important goal.

3.6.6.3 Performance Guidelines

<u>Planning and Implementation</u>

Although the risk of adverse impacts is limited, the successful use of whole lake circulation or hypolimnetic aeration requires a thorough study of the lake to determine if the method is likely to succeed, what type of treatment to employ, and the size and type of pumps required. Unacceptable results have routinely been traced to inadequate equipment or operation thereof, although chemical features of the target lake may also be responsible. Whole lake circulation requires an air compressor and pipes (usually metal for the first 30 meters to prevent damage, thereafter cheaper plastic pipe can be used if properly weighted). Plastic pipes can also be easily perforated to create a diffuser in the deepest part of the lake, with care taken to suspend the diffuser section about one meter above the bottom. Scuba divers may be required to install pipes and the diffuser.

Calculations can be performed to determine if the mixed depth will exceed the critical depth where light limitation is predicted to result in zero net production of algae. Descriptions on the use of these calculations are summarized in Cooke et al. (1993). A discussion of aerator sizing, aeration efficiency and general theory is presented in Kortmann et al. (1994). Bubble size and distribution and overall air delivery rate are important considerations for diffusers, and some careful engineering will greatly improve efficiency

Hypolimnetic aeration can be complicated to apply, given the need to deliver and distribute the appropriate amount of oxygen without destroying stratification. The size of the pumps is generally smaller than that required for whole lake circulation, but effectiveness is linked to oxygen transfer, not mixing. Calculating the transfer of oxygen is a technical task, but a general rule is that 2.5% of the oxygen is transferred for each vertical meter of contact. As most Massachusetts lakes have hypolimnia of much less than 10 meters thickness, this suggests that only a small part of the injected oxygen will be transferred. The chamber for partial lift systems must be carefully designed and use of a full lift system must avoid mixing hypolimnetic water with the epilimnion. Generally this technique is suggested only for lakes with large hypolimnetic volumes, relative to the epilimnetic volume, as exchange between layers will be more influential in these cases.

Caution should be exercised with aeration and mixing in lakes during winter, as these techniques may cause thin ice and dangerous conditions. In some cases the prevention of ice formation is desired, as with marina areas and some structures in northern lakes. Holes or areas of thin ice can occur, however, and represent a hazard for ice use by winter recreation enthusiasts.

<u>Monitoring and Maintenance</u>

In addition to electricity, the compressor or other machinery will require maintenance as specified by the manufacturer. Frequent monitoring of oxygen concentrations and temperature at various depths in the lake may be required to determine the minimum pumping rate required for each method. Additional biological monitoring should be conducted to determine how the algae, zooplankton and fish have responded to the treatment, and to suggest ways of regulating circulation or hypolimnetic aeration to maximize benefits and minimize costs. Although careful design will surmount most problems, expect to make adjustments to optimize performance.

Equipment failure and vandalism have been the most commonly reported maintenance issues. The artificial circulation of Lake Cochituate in Natick in 1971 and 1972 failed to control algae because of equipment failure, vandalism, and failure to monitor and maintain equipment (Cortell and Associates, 1973). Well maintained systems should operate for at least a decade, however, and some have been in use that long with limited parts replacement.

<u>Mitigation</u>

Other than mitigative measures to ensure that the diffusers are properly suspended above the sediments to reduce the possibility of sediment resuspension, the primary mitigative measure is to shut the system off if it is not working properly. It may be necessary to run the system for the whole growing season, not just to maintain desirable conditions, but to mitigate problems that may result from shutting it off (e.g., the fish kill in Mt. Williams Reservoir, North Adams).

3.6.7 Regulations

3.6.7.1 Applicable Statutes

In addition to the standard check for site restrictions or endangered species (Appendix II.), several permits may be applicable. A Notice of Intent must be sent to the Conservation Commission with a copy to the Department of Environmental Protection Regional Office. If the proposed project occurs within an Estimated Habitat of Rare Wildlife in the most recent version of the Natural Heritage Atlas, a copy of the Notice of Intent must be submitted to the Natural Heritage and Endangered Species Program (NHESP) within the MDFG for review (Appendix II). If the proposed project occurs within a Priority Habitat of Rare Species in the most recent version of the Natural Heritage Atlas, the project proponent must submit project plans to the NHESP for an impact determination. An Order of Conditions must be obtained prior to work. A Chapter 91 Permit may be required for installation of equipment (Appendix II) in Great Ponds. Small privately owned ponds may only require a Negative Determination of Applicability from the Conservation Commission.

3.6.7.2 Impacts Specific to Wetlands Protection Act

The following overall impact classification is offered as a generalization of impacts, with clarifying notes and caveats as warranted:

1. Protection of public and private water supply – Benefit (water quality improvement).
2. Protection of groundwater supply – Neutral (no significant interaction).
3. Flood control - Neutral (no significant interaction).
4. Storm damage prevention – Neutral (no significant interaction).
5. Prevention of pollution - Benefit (water quality enhancement).
6. Protection of land containing shellfish - Benefit (through water quality enhancement) with rare detriment by water quality variability induced by whole lake circulation.
7. Protection of fisheries - Benefit (through water quality enhancement) with rare detriment by water quality variability and loss of coldwater habitat induced by whole lake circulation.
8. Protection of wildlife habitat – Benefit (water quality enhancement).

Adverse impacts to the eight interests of the Wetland Protection Act are not expected with the exception that in rare cases deleterious substances like hydrogen sulfide or ammonia may be circulated to the surface and cause temporary adverse impacts to fish and wildlife. In general, aeration is expected to improve habitat for fish and other organisms in lakes with anoxic hypolimnia, but artificial circulation can reduce or eliminate coldwater habitat for trout. If the management project is successful at reducing nutrients and algal blooms in the lake there may be long-term benefits to some of the interests of the Wetlands Protection Act.

3.6.8 Costs

3.6.8.1 Whole Lake Circulation

Costs include the initial purchase and installation of the pumps, pipes and diffusers as well as annual maintenance costs and annual electricity costs. A review of numerous projects suggests initial costs range from about $365 to $4,292/ha (median $905/ha or $367/acre), and annual costs range from $108 to $2,068/ha (median $403/ha or $164/acre) in 2000 dollars (Cooke et al., 1993a). Actual costs depend on the amount of air required, which is related to lake area. Wagner

(2001) indicates an all-inclusive cost range of \$300 to \$5,000/acre for circulation systems in 2000 dollars, with an estimated range for 20 years of application at a hypothetical 100-acre lake of \$70,000 to \$400,000.

3.6.8.2 Hypolimnetic Aeration

Cooke et al. (1993) report costs on a per kg oxygen basis as approximately \$2.50/kg O_2 with operating costs of \$0.072/kg O_2. Assuming a need to counteract an oxygen demand of 500 to 2000 mg/m2/day for 120 days per year, this suggests a capital cost of \$756 to \$3024/acre and an annual operational cost of \$55 to \$218/acre in 2000 dollars. Wagner (2001) indicates an all-inclusive cost range of \$500 to \$3,000/acre for circulation systems in 2000 dollars, with an estimated range for 20 years of application at a hypothetical 100-acre lake of \$120,000 to \$400,000.

The layer aeration system for Lake Shenipsit in Connecticut cost approximately \$180,000 to install (\$340/acre) in the early 1990s and costs approximately \$15,000-20,000 per year (\$28-\$38/acre/year) to operate (R. Kortmann, ECS, pers. comm., 1996). Another layer air system cost \$280,000 (\$2,545/acre) in the mid-1990s and is used to aerate the 110-acre Bear Creek Reservoir in Denver (R. Kortmann, ECS, pers. comm., 1996).

3.6.9 Future Research Needs

Additional review of the existing cases should be conducted to try to explain what variable (e.g. nutrient concentration of hypolimnion vs. epilimnion, critical depth for light limitation, etc.) can best be used to predict the ecosystem response to circulation. Pastorok and Grieb (1984) used multiple discriminate analysis to examine how such variables as aeration rates, lake area, volume and depth could be used to predict management success. However their models correctly predicted success or failure in only 67 to 85 percent of the cases, and many comparisons were not statistically significant. Future implementation and statistical models should include detailed data on before and after concentration profiles for nutrients, temperature, oxygen, pH, dissolved CO_2, algae and zooplankton so that the effective mechanisms can be determined in each case. Impacts of aeration on fisheries could use some additional investigation, mainly as a function of monitoring programs for lakes with aeration.

3.6.10 Summary

Artificial circulation and hypolimnetic aeration offer potential for reduced phosphorus and algal abundance by minimizing internal recycling and fostering better zooplankton habitat through oxygen addition. Artificial circulation may also disrupt blue-greens by physical mixing and impact a wider range of algae through variation in light and shifts in pH brought on by mixing. Typically there are significant improvements in hypolimnetic water quality, and increases in oxygenated habitat for zooplankton, fish and other organisms. However, actual reductions in phosphorus concentration and algal abundance have not been consistent or reliable. Phosphorus declines in lakes with hypolimnetic aeration by one to two thirds on average, while artificial circulation has caused increased phosphorus levels about as often as it has decreased fertility. These techniques are commonly applied in drinking water reservoirs, where management of deep water quality is often an important consideration. These methods have also been popular in small ponds, where effectiveness in reducing phosphorus availability and algal abundance has not been clearly documented, but where mixing does tend to improve the visual appeal of the pond even with no change in algal biomass. There have been relatively few applications to recreational lakes, but these techniques are applicable.

There are few if any adverse effects expected from hypolimnetic aeration. Whole lake circulation may cause adverse impacts if the bottom waters are nutrient-rich compared to surface waters, and the technique may eliminate coldwater fisheries as the lake is mixed. Both methods can be controlled to minimize negative impacts by adjusting air flow or shutting them off. It is difficult to predict if these methods will control algae or cause a shift to more desirable species in any individual case without some experimentation. Knowledge gained over the last decade has improved system design and may allow more effective use of these techniques in the future. Combination with phosphorus inactivators, especially iron, has produced positive results.

3.7 DREDGING

3.7.1 Overview

Dredging is perhaps best known for maintaining navigation channels in rivers, harbors and ports or for underwater mining of sand and gravel, but dredging can also be an effective lake management technique for the control of excessive algae and invasive growth of macrophytes (Holdren et al., 2001). The management objectives of a sediment removal project are usually to deepen a shallow lake for boating and fishing, or to remove nutrient rich sediments that can cause algal blooms or support dense growths of rooted macrophytes. Dredging is discussed here in its role as a nutrient control strategy, but the discussion of available approaches is relevant to later discussion of macrophyte controls (Section 4).

The release of algae-stimulating nutrients from lake sediments can be controlled by removing layers of enriched materials. This may produce significantly lower in-lake nutrient concentrations and less algal production, assuming that there has been adequate diversion or treatment of incoming nutrient, organic and sediment loads from external sources. Even where incoming nutrient loads are high, dredging can reduce benthic mat formation and related problems with filamentous green and blue-green algae, as these forms may initially depend on nutrient-rich substrates for nutrition. Dredging also removes the accumulated resting cysts deposited by a variety of algae. Although recolonization would be expected to be rapid, changes in algal composition can result.

Dredging can be accomplished by multiple methods that can be conveniently grouped into five categories:

- Dry excavation, in which the lake is drained to the extent possible, the sediments are dewatered by gravity and/or pumping, and sediments are removed with conventional excavation equipment such as backhoes, bulldozers, or draglines.
- Wet excavation, in which the lake is not drained or only partially drawn down (to minimize downstream flows), with excavation of wet sediments by various bucket dredges mounted on cranes or amphibious excavators.
- Hydraulic dredging, requiring a substantial amount of water in the lake to float the dredge and provide a transport medium for sediment. Hydraulic dredges are typically equipped with a cutterhead that loosens sediments that are then mixed with water and transported as

pumped slurry of 80 to 90% water and 10 to 20% solids through a pipeline that traverses the lake from the dredging site to a disposal area.
- Pneumatic dredging, in which air pressure is used to pump sediments out of the lake at a higher solids content (reported as 50 to 70%). This would seem to be a highly desirable approach, given containment area limitation in many cases and more rapid drying with higher solids content. However, few of these dredges are operating within North America, and there is little freshwater experience upon which to base a review. Considerations are much like those for hydraulic dredging, and pneumatic dredging will not be considered separately from hydraulic dredging for further discussion.
- Reverse layering, which is grouped with dredging because it involves the movement of sediment, but differs in that the sediment is not actually removed from the lake. Sandy substrates beneath layers of muck are pumped upward and spread over the muck, burying the nutrient-rich material and creating a new top layer of presumably low-nutrient sand.

Dry, wet and hydraulic methods are illustrated in Figure 3-2. Cooke et al. (1993) provides a discussion of dredging considerations that will be helpful to some readers. Recent developments, methods, impact assessment and methods for handling dredged material can be found in McNair (1994). No technique requires more up front information about the lake and its watershed, and there are many engineering principles involved in planning a successful dredging project. No technique is more suitable for true lake restoration, but there are many potential impacts that must be considered and mitigated in the dredging process. Failed dredging projects are common, and failure can almost always be traced to insufficient consideration of the many factors that govern dredging success.

A properly conducted dredging program removes accumulated sediment from a lake and effectively sets it back in time, to a point prior to significant sedimentation. Partial dredging projects are possible, but for algal control it is far better to remove all nutrient-rich sediment, as interaction between sediments and the water column in one area can affect the entire lake. Many benefits beyond algal control are accrued from a proper dredging of a lake, including increased water depth, control of rooted plant growths, and reduced sediment-water interactions. The cost of dredging is often prohibitive, however, and an investment in dredging should be protected by an active watershed management program.

While removing nutrient-rich layers of sediment can control algae, dredging is most frequently done to deepen a lake, remove accumulations of toxic substances, or to remove and control macrophytes. Algal control benefits are largely ancillary in these cases. The expense of sufficient soft sediment removal, the alternative afforded by phosphorus inactivation, and the more pressing need for watershed management in most cases are the primary reasons that dredging is not used more often for algal control.

3.7.2 Dry Dredging

Dry dredging involves partially or completely draining the lake and removing the exposed bottom sediments with a bulldozer or other conventional excavation equipment and trucking it away. In general, small projects (< 30,000 cubic yards, or cy) involving silts, sands, gravel and larger obstructions and possessing manageable water level controls (i.e., pond drains), favor conventional, dry methodology (C. Carranza, BEC, pers. comm., 1996). Although ponds rarely dry to the point where equipment can be used without some form of support (e.g., railroad tie mats or gravel placed to form a road), excavating under "dry" conditions allows very thorough sediment removal and a complete restructuring of the pond bottom. Even without convenient water level control, pumping is sometimes employed to create the driest conditions possible. Short-term impacts will often be high, unless the pond is divided into sections and dewatered and refilled sequentially, but the long-term benefits of complete restoration are prized where the habitat has become severely degraded.

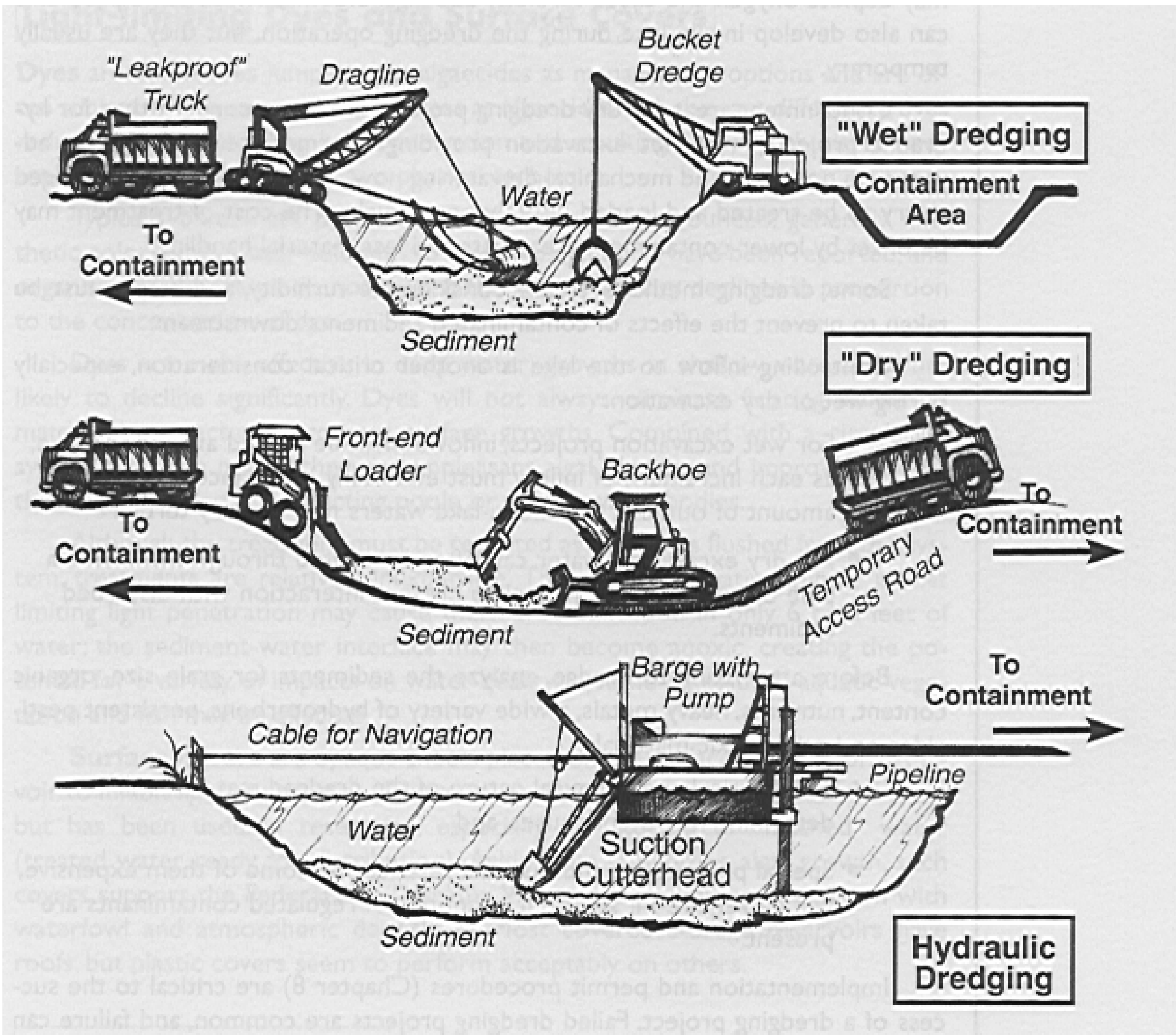

Figure 3-2 Dry, wet and hydraulic dredging approaches (from Wagner, 2001).

Control of inflow to the lake is critical during dry excavation. For dry excavation, water can often be routed through the lake in a sequestered channel or pipe, limiting interaction with disturbed sediments. Water added from upstream or directly from precipitation will result in solids content rarely in excess of 50% and often as low as 30%. Consequently, some form of containment area is needed before material can be used productively in upland projects. Where there is an old gravel pit or similar area to be filled, one-step disposal is facilitated, but most projects involve temporary and permanent disposal steps.

3.7.3 Wet Dredging

Wet dredging may involve a partial drawdown, especially to avoid downstream flow of turbid water, but sediment will be excavated from areas overlain by water in such projects. Such sediment will be very wet, often only 10 to 30% solids unless sand and gravel deposits are being removed. Clamshell dredges, draglines, and other specialized excavation equipment are used in wet dredging operations. Excavated sediment must usually be deposited in a bermed area adjacent to the pond or into tight tanks or other water-holding structures until dewatering can occur. This approach may be necessary in association with dry dredging projects when water level control is not complete, and is most often practiced on only a small scale (<10,000 cy). An exception is harbor dredging, as has been practiced in Boston and New Bedford, where large quantities of sediment are removed by wet dredging since no water level control is possible.

Conventional wet dredging methods create considerable turbidity, and steps must be taken to prevent downstream mobilization of sediments and associated contaminants. For wet excavation projects, inflows must normally be routed around the lake, as each increment of inflow must be balanced by an equal amount of outflow, and the in-lake waters may be very turbid. It should be noted, however, that more recent bucket dredge designs greatly limit the release of turbid water and have been approved for use in potentially sensitive aquatic settings.

3.7.4 Hydraulic (and Pneumatic) Dredging

A more advanced form of wet dredging, hydraulic dredging usually involves a suction type of dredge that has a cutter head. Agitation combined with suction removes the sediments as a slurry containing approximately 15-20 percent solids by volume, although this may increase to as high as 30 to 40 percent in some cases or be as low as 5% with especially watery sediments in difficult areas. This slurry is typically pumped (or barged in rare cases) to a nearby containment area on shore where the excess water can be separated from the solids by settling (with or without augmentation). The supernatant water can be released back to the lake or some other waterway. The containment area for a hydraulic dredging project is usually a shallow diked area that is used as a settling basin. The clarified water may be treated with flocculation and coagulation techniques to further reduce the suspended solids in the return water (Church et al., 1994).

Hydraulic dredging is normally favored for removal of large amounts of sediments, particularly highly organic sediments with few rocks, stumps or other obstructions and where water level control is limited (C. Carranza, BEC, pers. comm., 1996). This type of project does require a containment area to be available where removed sediments are separated from water, and may involve secondary removal of the dried sediment from the containment area for ultimate disposal elsewhere. Usually the containment area is not far from the lake, but in several cases the slurry has been pumped as far as 8 miles to a suitable disposal area (Weathersbee, C&B, pers. comm., 1998).

Innovations in polymers and belt presses for sediment dewatering have reached the point where a hydraulically dredged slurry can be treated as it leaves the lake to the extent necessary to load it directly onto trucks for transport to more remote sites. Solids content of the resultant material is still too low for many uses without further drying or mixing with sand, but the need for a large containment area can be avoided with this technology. The cost of coagulation and mechanical dewatering may be at least partially offset by savings in containment area construction and ultimate material disposal.

3.7.5 Reverse Layering

An alternative method to dredging that is believed to provide some of the same benefits is the reverse layering of sediments. It is a still largely experimental procedure that is being tested in small areas of Red Lily Pond in Barnstable, Massachusetts. It is believed to be especially applicable to the glacial "kettle hole" ponds that are common to Cape Cod and Southeastern Massachusetts because of a layer of glacial sand that lies beneath the accumulated muck layer. The purpose is to extract glacial sand that underlays the nutrient-rich, anaerobic, organic sediments of a eutrophic lake and place it on top of those less desirable sediments.

Reverse layering is accomplished by hydraulic jetting. Water is pumped down below the muck and/or peat layer to the deep layer of glacial sand. The glacial sand is forced up through pipes and spread over the bottom sediments. A cavity is created by the removal of glacial sand, which causes the bottom sediments to subside and fill the cavity. The purpose of this method is to retard or reverse the process of eutrophication, and to restore the lake bottom to the original sediment type that will promote a more diverse plant and animal community (K-V Associates, Inc., 1991). This method does not require disposal of dredge materials, nor does it deepen the lake. It simply switches the location of existing sediment layers.

Reverse layering is not considered dredging by some groups, most notably the US Army Corps of Engineers, which therefore does not require a permit under Section 404 of the Clean Water Act as it normally does for dredging projects. It is certainly a very different technique than the other methods of actual sediment removal, but the underlying goal is the same with regard to nutrient and algae control; limit the availability of nutrients from accumulated muck sediments.

3.7.6 Effectiveness

Dredging can be a very successful lake management technique for reducing the occurrence of algal blooms under the right circumstances. Dredging controls nutrients by removing nutrient-rich sediments and increasing depth. The increase in depth may lessen the occurrence of summer turnover in shallow lakes and slow the release of sediment phosphorus (see Stephan and Hanson, 1980). Increased volume allows greater dilution of nutrients from a stable or reduced load. Immediate post-dredging results are usually striking. However, for this technique to have long-term effectiveness, methods for control of nutrients entering the lake from external sources must be implemented.

3.7.6.1 Short-Term

Immediately after dredging, with most or all soft sediment removed, release of nutrients back into the water column will be minimized and desired results should be maximized. If sediments were not the primary nutrient source, results may be less dramatic, but usually the short-term results of dredging are quite acceptable in terms of water quality and algal abundance. Given the cost and effort involved, however, short-term effectiveness is not as critical as long-term effectiveness.

3.7.6.2 Long-Term

Sediment removal to retard nutrient release has been effective in documented cases. An example is provided by Lake Trummen in Sweden (Andersson 1988) where the upper 3.3 feet of sediments were extremely rich in nutrients. This layer was removed and the total phosphorus concentration in the lake dropped sharply and remained fairly stable. The dredging began in 1970 when one half meter of sediment was removed, followed by an additional half meter the following year. Between 1968 and 1973 the Shannon diversity index of phytoplankton rose from 1.6 to 3.0, and Secchi disk transparency rose from 23 to 75 cm, indicating a more diverse algal population and increased transparency. The benthic community was recolonized within a year, with a slight change in species composition. Dredging greatly increased the quality of Lake Trummen both ecologically and recreationally. There has been little deterioration of the lake quality over the past 20 years (Cooke et al., 1993a; Peterson, 1982).

Algal abundance also decreased and water clarity increased in Hills Pond in Massachusetts after all soft sediment was removed and a storm water treatment wetland was installed in 1994 (K. Wagner, ENSR, pers. obs., 1996). Dredging of 6-acre Bulloughs Pond in Massachusetts in 1993 has resulted in abatement of thick green algal mats for eight years now, despite continued high nutrient loading from urban runoff (K, Wagner, ENSR, pers. obs., 1994-2001). These mats had previously begun as spring bottom growths, then floated to the surface in mid-summer.

The effectiveness of a sediment removal project as a nutrient control strategy depends largely on the pre-dredging assessment of the problem, the amount of sediment that is removed, and control of external nutrient and sediment loading. If any of these factors are inaccurately assessed, the treatment may be less successful. An example of this is Lake Henry, WI, where the infilling rate was severely underestimated, resulting in an underestimation of the post dredging sedimentation rate (Cooke et al., 1993a). The list of failed dredging projects is long, but in nearly all cases the failure is related to incomplete application of the technique or an incomplete lake management program. Dredging can correct past sedimentation impacts on lakes, but does not prevent future inputs.

The dredging of Liberty Lake in Washington was not successful at reducing the available phosphorus in the sediments because of the non-uniform trenching pattern used for dredging. Instead of removing all of the surface sediments, the contractor removed all sediments in spaced trenches and this allowed undredged adjacent surface sediments to slump into the trenches. The lake was deepened but not all of the phosphorus rich surface sediments were removed in the dredged areas. This demonstrates the importance of communicating to the contractor the purpose of the operation (to remove phosphorus rich surface sediments, rather than simply to make the lake deeper) (Moore et al., 1988).

The dredging at Lake Trehorningen in Sweden used a hydraulic dredge to move the loose topmost half-meter of sediment into a sequestered bay of the lake that was used as the settling pond. The effluent was treated by flocculation and chemical coagulation before return to the lake. The dredging treatment reduced total phosphorus in the lake by 50 percent (from about 436 µg/l to 226 µg/l in the eastern basin). Yet the phosphorus levels were still too high to see any decrease in algae chlorophyll a and the Secchi disk transparency remained at about 0.4 meters (Ryding, 1982). This poor response is to be expected when phosphorus levels are very high.

Dry dredging of Dunn Pond in Gardner successfully reduced profuse growths of macrophytes, but also eliminated a thick layer of nutrient-rich muck that had influenced water quality. The post-dredging bottom was coarse sand, gravel and placed rock (for habitat). A detention system with a filter berm was installed to clean incoming waters. The post-restoration chlorophyll a concentrations were mostly less than 3 µg/L and the lake is considered to be oligotrophic; the lake now supports a put-and-take trout fishery and extensive recreational opportunity in the form of boating and swimming (MDEP, 1994)

Reverse layering was shown to be effective in the short-term in Red Lily Pond for macrophytes, but no assessment of water quality effects has been conducted. Laboratory work suggests that reduced phosphorus release and algal growth should result. As work is still progressing at Red Lily Pond, and only small areas have been treated, it is difficult to assess long-term effectiveness of this technique in terms of water quality throughout a lake.

3.7.7 Impacts to Non-Target Organisms

The risk of negative impacts by dredging on the lake and surrounding area is a function of the type of dredging, project design, and project implementation. Many possible problems are short-lived, however, and can be minimized with proper planning. It should be kept in mind that dredging represents a major re-engineering of a lake, and should not be undertaken without clear recognition of its full impact, positive and negative. Impacts to non-target organisms are discussed here by type of dredging.

3.7.7.1 Short-Term Impacts of Hydraulic (and Pneumatic) Dredging

Since the dredge can only operate in a small area at any one time and the lake is maintained as an aquatic habitat during dredging, short-term impacts to mobile species are minimal and impacts to non-mobile organisms are localized. A reduction of benthic dwellers, a decrease in benthic fish food, loss of habitat for benthic dwellers, and removal of non-target aquatic plants are all expected in hydraulically dredged areas as the sediment and any associated biota are removed. If the sediments are anoxic, few biota will be present, but in shallow areas the impact to non-target flora and fauna can be substantial. Even if the biota are not directly impacted, the intent of such dredging is usually to change the nature of the substrate, so conditions inhospitable to pre-existing biota would be expected. Recolonization of dredged areas is usually gradual and the new community may represent an improvement on pre-dredging biota, so dredging impacts on benthic biota are often considered acceptable. Where protected species or other biota of interest are involved, impacts should be considered on a lakewide basis to determine if dredging is an acceptable approach.

Impacts associated with sediment resuspension, high turbidity, release of nutrients and toxic substances from the sediments and lowered dissolved oxygen concentrations have been postulated but are rarely an issue for hydraulic dredging. Disturbed sediments are sucked into the pipeline and turbidity is rarely above ambient background for the lake outside of 10 to 20 ft from the cutterhead. The potential for impacts to fish eggs or fry by siltation and smothering during spawning periods is minimal, although actual dredging of spawning areas during or shortly after spawning could certainly cause impacts.

Increases in nutrient concentrations are possible as a function of mixing of sediment and water when the water is later returned to the lake. Coagulation of the dredged slurry (often with alum) to sufficiently clarify it before discharge back to the lake (or other waterbody) should reduce nutrient levels to an acceptable level, but some impact is possible. Any resultant increase in phytoplankton productivity is generally short-lived (Cooke et al., 1993a; Olem and Flock, 1990; Peterson, 1982). Likewise, the release of other contaminants into the water during mixing with sediment in the pipe and containment area could result in a contaminated discharge to the lake, but proper operation of the containment area will prevent such impacts. Impacts observed in past dredging projects have typically been related to improper design or operation. It is also possible to cause lake drawdown if pumping rates are high relative to inflow rates and the overflow from the containment area is not returned to the lake.

There are possible impacts associated with the deposition of sediments in the containment area as the habitat is buried under large amounts of wet sediments. However, containment areas permitted in Massachusetts for hydraulic dredging are usually highly engineered and disturbed upland sites where no biota of concern would be present at the time of dredging. Restoration of the containment area usually results in habitat enhancements.

3.7.7.2 Long-Term Impacts of Hydraulic (and Pneumatic) Dredging

The impacts to benthic organisms are generally expected to be short-term. Benthic organisms are able to move from other undisturbed areas of the lake and recolonize the dredged area. If the lake is completely dredged, re-establishment of benthic fauna could take 2 to 3 years (Cooke et al., 1993a), and may involve a community of different composition. Dredging is typically conducted in degraded habitats of low species diversity that dredging may improve in the long run. Potential long-term impacts to fisheries and other wildlife are largely a function of altered habitat, and for most dredging projects, habitat is considered to be improved for the majority of species. Still, the intended change in bottom conditions will represent a negative change for some species. In a review of several dredging studies Cooke et al., (1993) suggests most impacts are short lived and generally acceptable relative to long-term benefits of the technique.

Impacts at the containment area and at the final disposal site are due to direct impact of burial under wet sediments and possible contamination of soil and groundwater if toxins or heavy metals are present in the dredged material. Impacts can be mediated to some extent by spreading material in thin layers, mixing with other soils, and burial under other material. However, where a real threat of contamination exists, dredged material disposal is tightly regulated and may make dredging uneconomical.

3.7.7.3 Short-Term Impacts of Dry Dredging

Because the lake is drained and much of the bottom scraped, widespread impact to non-mobile and water dependent species are expected in the short-term. Short-term impacts to non-target organisms would include impacts listed above for wet dredging, plus those listed for lake drawdown in Section 4. Wildlife that get food from the lake may find conditions during dredging advantageous, as food items are exposed, but this short-term benefit may become a disadvantage if the dredging project is prolonged.

Dry dredging is generally undertaken to restore a degraded habitat to a former condition, and would not typically be recommended or approved if protected species are present or the biological community is considered highly desirable. Where dry dredging is proposed, some evaluation of recolonization should be provided, or an active re-population program should be proposed. Algae and invertebrates rarely need any help in recolonizing an area, but fish and rooted plant introductions may be desirable. Where the water body is isolated, natural recolonization will take longer than for lakes connected to other water bodies by streams. If the lake is only partially drawn down to gain access to nearshore areas for partial dredging, less impact is expected. Dry dredging represents a major overhaul of lake conditions, and can greatly improve conditions for the future where properly applied, but short-term impacts may be substantial and unavoidable.

Deposition of dredged material in a containment area and/or ultimate disposal area involves less water than for hydraulic dredging, and the prepared area is usually devoid of species of concern. Consequently, quality of return water is not usually an issue, except possibly for groundwater near the containment site, and impacts to biota in the containment area are expected to be minimal.

3.7.7.4 Long-Term Impacts of Dry Dredging

The time needed for re-establishment of flora and fauna is dependent on to what extent the lake bottom is dredged and the proximity of sources of these biota (Cooke et al., 1993a). If some areas are left undisturbed, benthic organisms can recolonize the dredged areas if those areas are hospitable in their post-dredging condition. If the entire bottom is dredged, then benthic species as well as fish and other pelagic species may recolonize from appropriate refuge habitats either upstream or downstream of the lake. If areas are dredged to a very different condition, colonizing species may constitute a community quite different from that present before dredging (and this may be part of the intent of dredging). Restoration of all species could take years if no suitable refuge habitat is available nearby. Where dry dredging of an entire lake is planned, consideration of what biota are expected or desired is strongly encouraged. The value of dredging can be enhanced by selected introductions with follow-up monitoring and control of invasive species.

3.7.7.5 Short-Term Impacts of Wet Dredging

Wet dredging maintains some water in the lake like hydraulic dredging, but usually involves some degree of drawdown like dry dredging. Wet dredging does not typically control turbidity, so impacts associated with high turbidity, relocated siltation, drastically changed and variable water quality, and direct impacts on biota are all possible. Use of newer wet dredge designs may minimize turbidity, however. Wet dredging should generally be practiced only where the habitat is extremely degraded prior to dredging, and active restoration of habitat and desirable communities is planned. Short-term disruption of populations in the lake is likely, but wildlife that get food from the lake may find conditions improved during dredging as food items are disturbed or exposed. It is important to avoid downstream transport of the water in the lake during wet dredging, as downstream impacts would be probable. The containment area for a wet dredging project may be much like that for hydraulic dredging, and the quality of return water and eventual restoration of the containment area should be addressed in the planning stage of the project to minimize impacts.

3.7.7.6 Long-Term Impacts of Wet Dredging

Wet dredging projects tend not to be as complete as dry dredging, and may leave more material than a hydraulic dredging project. As such, recolonization to conditions more similar to pre-dredging conditions is likely, except where newly created depth limits growth of emergent macrophytes. Impacts during the wet dredging process may delay recovery, and the re-established communities will undoubtedly differ to some degree from pre-dredging biota (again, often by intent of the project), but long-term impacts from wet dredging are rarely adverse.

3.7.7.7 Short-Term Impacts of Reverse Layering

Short-term impacts are expected to be similar to those for hydraulic dredging, as water is maintained in the lake but the nature of the sediment is changed. Immobile benthic organisms are not removed, but may be buried to the point where impacts occur. The technique is often intended to reduce rooted plant growths by changing the substrate and burying existing populations. Turbidity may be higher than for hydraulic dredging, but will not be as high as for wet dredging with conventional excavation equipment, as the sand that is moved is less prone to resuspension than disturbed muck sediments. This technique has not been applied on a large enough scale to examine any lakewide effects.

3.7.7.8 Long-Term Impacts of Reverse Layering

Long-term impacts are expected to be similar to those listed for hydraulic dredging, although there is insufficient evidence to make any reliable predictions at this time.

3.7.8 Impacts to Water Quality

Dredging may impact water quality by removing sediment and reducing the interactions between sediment and water (usually a goal of the project) or by resuspending sediment and increasing the interaction between sediment and water (generally a negative impact to be avoided). Impacts to water quality vary by dredging type and are discussed by technique below.

3.7.8.1 Short-Term Impacts of Hydraulic Dredging

The sediment removal process can cause a short-term increase in turbidity on a localized basis, but widespread impact should not occur if equipment is functioning properly. Failure to properly settle and treat the slurry (if necessary) before discharge from the containment area presents a substantial risk of impact, but is avoidable.

The dredging of Lake Springfield, a very large (1,635 hectare = 4,038 acre) reservoir in Illinois was subject to intense opposition until it was demonstrated that the sediments were not significantly contaminated by pesticides. Large scale settling tests of the sediments were conducted in accordance with the U.S. Army Corps of Engineers Technical Report DS-78-10 "Guidelines for Design, Operating and Managing Dredging Material Containment Area" as cited in Buckler et al. (1988). Two oversized settling ponds (160 acre and 72 acre) with a retention period of 8.7 days were constructed to meet the 15 mg/L total suspended solids requirement for the effluent (Cooke et al., 1993a). Although the ammonia concentrations were as high as 25 mg/L in the slurry, no problems in the return effluent were reported (Buckler et al., 1988).

The hydraulic dredging of Liberty Lake in Washington resulted in minimal and transitory impacts to water quality. A sediment plume was created near the auger but did not affect transparency near the surface and changes in total solids, total phosphorus and chemical oxygen demand during the dredging operation were negligible (Breithaupt and Lamb, 1983). Similarly, there have been few instances of any turbidity or nutrient problems associated with dredging in southern New England (K. Wagner, ENSR, pers. obs., 1985-2002).

3.7.8.2 Long-Term Impacts of Hydraulic Dredging

The long-term effects of dredging on water quality are usually expected to include an increase in water clarity. If dredging removes organic sediment and leaves inorganic sediment as the new bottom, then there will be less release of nutrients from the bottom and less potential for resuspension by wind action. If dredging is incomplete in a large portion of the lake, this benefit may be compromised, but dramatic improvement in water quality is possible where soft sediments are completely removed. This is most often accomplished in dry dredging projects, although all forms of dredging have the potential to provide this benefit.

3.7.8.3 Short-Term Impacts of Dry Dredging

Water quality impacts should be limited by the absence of water during dry dredging, although complete control of water is rare. As the lake is drawn down during dry dredging, impacts associated with lack of water are generally of greater concern than water quality impacts.

3.7.8.4 Long-Term Impacts of Dry Dredging

An increase in water clarity is expected over the long-term. For thorough dry dredging projects, dramatic improvement in water quality is possible, with lower nutrients and solids and more stable dissolved oxygen and pH.

3.7.8.5 Short-Term Impacts of Wet Dredging

Bucket dredges and drag lines can generate substantial suspended solids levels in the lake, with average levels less than 200 mg/L for watertight buckets and less than 300 mg/L for open buckets (Mongomery, 1984). Some types of conventional wet excavation equipment are more specialized to reduce suspended solids and are useful for dredging contaminated sediments. Greatly increased interaction between sediments and the water column is expected, however, and increases in water column concentrations for any contaminants present are possible.

3.7.8.6 Long-Term Impacts of Wet Dredging

As with other forms of dredging, improved water quality is expected as a consequence of removal of soft sediment. However, unless the lake is "overdredged" (material removed beyond the soft sediment layer) or coarse material is added after dredging (capping, as performed in

Boston Harbor), at least a fine layer of soft sediment is likely to remain and may reduce water quality benefits.

3.7.8.7 Short-Term Impacts of Reverse Layering

Readings taken during reverse layering in Red Lily Pond showed a slight increase in turbidity at the lake outflow (0.9 to 2.0 NTU), and a decrease in pH, soluble phosphorus, total phosphorus, and total nitrogen concentrations. The decrease in phosphorus and nitrogen concentrations indicates that the nutrient rich, "mucky" top sediments were not releasing nutrients to the water column (K-V Associates, Inc., 1991). Insufficient information is available for further evaluation.

3.7.8.8 Long-Term Impacts of Reverse Layering

Water clarity is expected to improve over the long-term by reducing interaction of organic sediments with the water column, although no data are available to support this assumption.

3.7.9 Impacts in the Disposal Area

All dredged sediments must have a disposal area. The ideal situation is the use of the dredged material to reclaim damaged upland parcels, like old gravel pits. Some dredged material can also be used as cover for landfills, and especially clean material can be applied to agricultural fields or mixed with sand to make a topsoil-like fill used in landscaping. In rare cases, contractors want the dredged material and its value can partially offset the cost of dredging, but more often it is necessary to find a disposal site for unwanted dredged material. In most cases it is necessary to have both a temporary containment area in which dredged material is dewatered and a permanent disposal location (or multiple locations). Although many disposal arrangements have been made in the past, current regulations tightly control the manner in which dredged material is handled.

The primary regulations governing the disposal of lake sediments are the Massachusetts Contingency Program and various MDEP regulations and policies intended to minimize impacts from disposal (e.g., Interim Policy #COMM-94-007, Interim Policy for Sampling, Analysis, Handling and Tracking Requirements for Dredged Sediment Reused or Disposed at Massachusetts Permitted Landfills). Table 3-3 lists many of the thresholds relevant to dredged material disposal. The average concentrations for selected metals in Massachusetts lake sediments, from the many D/F studies conducted in the 1980s (Rojko, 1992), are also shown. It is particularly striking that the average level for most metals exceeds the 90th percentile for background soil conditions. This means that use in surficial landscaping will only be possible for lake sediments that are cleaner than average. Most urban lake sediments will not be so clean, although some are. Metals and hydrocarbons (especially benzene compounds) are most often problematic in urban lake sediments. Agricultural sediments may contain components of pesticides (e.g., arsenic, DDT derivatives) that exceed these thresholds.

Sediment quality issues are not unique to Massachusetts. The sampling of sediments from Hamlet City Lake in North Carolina revealed aliphatic and aromatic hydrocarbon contamination as well as elevated concentrations of some metals. A series of leaching tests was conducted and it was found that hydrocarbons should not be a problem, but the metals may adversely impact the groundwater (Brannon et al., 1993). Testing of sediments from Flint Pond in Hollis, NH in preparation for a possible dredging project revealed high levels of arsenic that may have come

Table 3-3 Massachusetts regulatory sediment quality values of importance to dredged material disposal. Mean lake and pond sediment metal values from MDEP D/F studies (Rojko, 1992) are included where available for comparison.

from nearby orchards many years ago. Although all other sediment features were favorable for disposal on agricultural fields, the arsenic levels were considered too high to proceed (K. Wagner, ENSR, pers. obs., 2002).

Sediment Quality Variable and Method	**MA Mean Lake and Pond Sediment Data (ppm)**	**MDEP Background Soil Data Set 90th Percentile (ppm)**	**MCP RCS-1. GW-1 (ppm)**	**Unlined Landfill Disposal Threshold (ppm)**
Metals (bulk chemistry)				
Aluminum		13,000		
Arsenic	17.1	16.7	30	40
Cadmium	4.6	2.06	30	30
Chromium (total)	23	28.6	1000	1000
Copper	41.8	37.7	1000	
Iron	16,692	17,000		
Lead	203	98.7	300	1000
Manganese	382	300		
Mercury	0.28	0.28	20	10
Nickel	23	17	300	
Zinc	195	116	2500	
Metals (TCLP)				
Arsenic				5
Cadmium				1
Chromium				5
Lead				5
Mercury				0.2
Polychlorinated Biphenyls			2	2
Pesticides				
Aldrin			0.03	
Chlordane			1	
DDT and derivatives			2	
Dieldrin			0.03	
Endosulfan/derivatives			20	
Endrin/Endrin aldehyde			0.6	
Heptachlor			0.1	
Heptachlor epoxide			0.06	

Extractable Petroleum Hydrocarbons				2500
C9-C18 Aliphatics			1000	
C19-C36 Aliphatics			2500	
C11-C22 Aromatics			200	

Table 3-3 Regulatory sediment quality values of importance to dredged material disposal. (continued)

Sediment Quality Variable	MA Mean Lake and Pond Sediment Data (ppm)	MDEP Background Soil Data Set 90th Percentile (ppm)	MCP RCS-1, GW-1 (ppm)	Landfill Disposal Threshold (ppm)
Polynuclear Aromatic Hydrocarbons				100
Acenaphthene			20	
Acenaphthylene			100	
Anthracene			1000	
Benzo(a)anthracene			0.7	
Benzo(a)pyrene			0.7	
Benzo(b)fluoranthene			0.7	
Benzo(k)fluoranthene			7	
Benzo(g,h,i)perylene			1000	
Chrysene			7	
Dibenzo(a,h)anthracene			0.7	
Fluoranthene			1000	
Fluorene			400	
Indeno(1,2,3-cd)pyrene			0.7	
Naphthalene			4	
Phenanthrene			100	
Pyrene			700	

Actual impacts on containment and disposal areas have not been well documented. As temporary containment areas are usually highly disturbed and engineered parcels, biotic impacts should be minimal. Possible leaching of contaminants into groundwater is one issue, and possible impacts on vegetation and biota once the containment area is restored may require consideration. Ultimate disposal impacts depend on the use of the material, with most regulations focused on preventing impacts to human health. Again, it is not clear that most contaminants are mobile enough or sufficiently reactive to cause ecological or human health impacts, but the potential for impact exists. Mitigation measures include prohibition on use within 500 ft of residences or in recreational settings, covering of dredged material with at least 18 inches of clean fill, and various blending schemes. Thin-layer disposal of dredge material has been promoted, as this is believed to reduce impacts to biota in the disposal area (Wilber, 1992).

3.7.10 Applicability to Saltwater Ponds

Dredging is as applicable to saltwater ponds as freshwater ponds. Additionally, the maintenance of openings between saltwater ponds and the ocean may be allowed in order to manage, maintain, or enhance marine fisheries. The applicant for a permit to conduct this type of project must show that the opening is for an approvable purpose, and that the conditions of the permit minimize adverse impacts to resource areas (DWW Policy 91-2 as printed in MDEP, 1995). The impacts to shellfish beds may be severe for any of the dredging operations described above, but could be an integral part of shellfishery restoration and maintenance as well.

3.7.11 Implementation Guidance

3.7.11.1 Key Data Requirements

A nutrient budget is needed to determine if removal of sediments will have a sufficient effect on nutrient levels to warrant dredging for that purpose. Dredging may well be undertaken for reasons of water depth and macrophyte control, but effectiveness as a nutrient control and algal management method depends upon the relative importance of internal and external nutrient loads. Data requirements for planning a successful dredging project are so extensive that a professional analysis is generally required. Table 3-4 summarizes most needs. Sediment quality is the most critical information need, followed by sediment quantity and containment/disposal area options.

Pre- and post- treatment biological, chemical and physical surveys should be conducted to assess impacts. The control of excess turbidity is often a critical concern in these types of treatments. For dry dredging this is a function of inflow control, while for wet dredging it may be difficult, although sequestering the dredged area may be possible. For hydraulic or pneumatic dredging, turbidity control is a function of containment area design and operation. Finally, estimates of effectiveness should be made for lake recovery in terms of total phosphorus levels and water transparency over the long-term.

3.7.11.2 Factors that Favor this Approach

The following considerations are indicative of appropriate application of dredging for reductions in nutrient concentrations and control of algae in lakes:

1. A substantial portion of the P load is associated with sediment sources within the lake.
2. Studies have demonstrated the impact of internal loading on the lake.
3. External P and sediment loads have been controlled to the maximum practical extent or are documented to be small; historic loading may have been much greater than current loading.
4. Sediments are “clean”, based on Massachusetts regulatory thresholds.
5. Suitable and sufficient containment and disposal areas are available close to the lake.
6. Additional goals of increased depth and/or macrophyte density reduction are important.
7. For conventional wet or dry dredging, habitat is degraded to the extent that a complete restructuring is desirable.
8. For conventional wet or dry dredging, partial drawdown or sequestering of the dredged area can be performed to limit impacts to aquatic species.
9. For hydraulic dredging, rocks, stumps and other obstructions are minimal.

Table 3-4 Key considerations for dredging

Reasons for Dredging:
Increased depth/access
Removal of nutrient reserves
Control of aquatic vegetation
Alteration of bottom composition
Habitat enhancement
Reduction in oxygen demand

Existing and Proposed Bathymetry:
Existing mean depth
Existing maximum depth
Proposed distribution of lake area over depth range
Proposed mean depth
Proposed maximum depth
Proposed distribution of area over depth range

Volume of Material to Be Removed:
In-situ volume to be removed
Distribution of volume among sediment types
Distribution of volume over lake area (key sectors)
Bulked volume (see below)
Dried volume (see below)

Physical Nature of Material to Be Removed:
Grain size distribution
Solids and organic content
Settling rate
Bulking factor
Drying factor
Residual turbidity

Nature of Underlying Material to Be Exposed:
Type of material
Comparison with overlying material

Dewatering Capacity of Sediments:
Dewatering potential
Dewatering timeframe
Methodological considerations

Chemical Nature of Material to Be Removed:
Metals levels
Petroleum hydrocarbon levels
Nutrient levels
Pesticides levels
PCB levels
Other organic contaminant levels
Other contaminants of concern (site-specific)

Protected Resource Areas:
Wetlands
Endangered species
Habitats of special concern
Species of special concern
Regulatory resource classifications

Flow Management:
System hydrology
Possible peak flows
Expected mean flows
Provisions for controlling water level
Methodological implications

Equipment Access:
Possible input and output points
Land slopes
Pipeline routing
Property issues

Relationship to Lake Uses:
Impact on existing uses during project
Impact on existing uses after project
Facilitation of additional uses

Potential Disposal Sites:
Possible containment sites
Soil conditions
Necessary site preparation
Volumetric capacity
Property issues
Long term disposal options

Dredging Methodologies:
Hydraulic (or pneumatic) options
Wet excavation
Dry excavation

Table 3.4 Key considerations for dredging (continued)

Applicable Regulatory Processes:
MEPA review (Environmental Notification Form)
Environmental impact reporting (EIR if needed)
Wetlands Protection Act (Order of Conditions)
Dredging permits (Chapter 91)
Aquatic structures permits (Chapter 91)
Water Management Act (diversion/use permits)
Clean Water Act Section 401 (WQ certification)
Clean Water Act Section 404 (USACE wetlands statute)
Dam safety/alteration permit (MDCR)
Waste disposal permit (MDEP)
Discharge permits (NPDES, USEPA/MDEP)

Removal Costs:
Engineering and permitting costs
Construction of containment area
Equipment purchases
Operational costs
Contract dredging costs
Ultimate disposal costs
Monitoring costs
Total cost divided by volume to be removed

Uses or Sale of Dredged Material:
Possible uses
Possible sale
Target markets

Other Mitigating Factors:
Necessary watershed management
Ancillary project impacts
Economic setting
Political setting
Sociological setting

Planning and Implementation

Dredging is an expensive technique, and while improvements can be spectacular, mistakes can be costly and impact-laden. Consideration of project feasibility, careful planning and anticipation of possible problems are crucial to success. Best management practices should be employed in the watershed to reduce nutrient and sediment loading before a dredging project with nutrient control as a goal is implemented, unless it is certain that the watershed is not a significant source of nutrients.

Before dredging is planned, it is important to conduct an analysis of the sediments for grain size, organic content, nutrients, heavy metals, a wide variety of hydrocarbons, persistent pesticides, and other potentially toxic or otherwise regulated materials. The physical and chemical nature of the dredged material will determine its potential uses and regulatory restrictions on its handling and disposal. Special precautions and disposal limitations, some of them expensive, will be required if certain substances are present above threshold concentrations. Implementation and permit procedures are critical to the success of a dredging project, and current project feasibility is controlled mainly by sediment quality.

Although dredging is rarely applied solely to control nutrients, an accurate nutrient budget including both a measured mass balance and a land-use source analysis should be conducted if nutrient control is a goal. Detailed analysis of internal sources of phosphorus, relative to external sources, is especially important. If the nutrient budget indicates the sediments as the major source of phosphorus, then dredging may be effective. Dredging may reduce the density of aquatic macrophytes and algae by removing nutrient-rich substrate or by increasing the depth, thereby inducing light limitation, and these are usually goals of dredging as well. However, to achieve lasting results for nutrient control, the nutrients removed with sediments must be the primary source of nutrients to the lake. Some of the general lake characteristics that indicate the applicability of sediment removal as a control method are expansive deposits of organic sediment, low sedimentation rates, and long hydraulic residence times.

Sediment quantity is almost as critical a consideration for dredging projects as sediment quality. It is possible to be successful while removing only a portion of the sediment if a low-nutrient layer can be exposed or the remaining sediment is not enough to have a major impact on water quality. However, most successful dredging projects target complete removal of nutrient-rich sediments, which is usually equated with all organic sediments. The depth of "soft" nutrient-rich sediments can be roughly determined by pushing a rod or pipe into the sediments until firm resistance is felt, usually indicating the depth to coarse sand, gravel or rock. Coring surveys (vertical samples of the sediments) on a smaller scale than sediment probing are vital to confirm sediment depth and characterize any sediment horizons as part of the pre-dredging planning (Moore et al., 1988). The use of ground-penetrating sound waves and related higher technology has been successfully employed in some cases, but confirmation with cores is still recommended. Incorrect assessment of soft sediment depth and underestimate of volume to be removed has been a problem for some past dredging projects, leading to either failure to achieve goals or greater expense than initially expected.

There are many factors to consider in choosing a containment area. The primary factors controlling containment area selection and design are the amount of material to be disposed, the ability to maintain required effluent quality, distance and access routes for getting sediment to the area, and the potential for restoration after disposal is complete. Among the most serious dredging problems is the failure to have a disposal area of adequate size to handle the necessary volume of sediment or turbid, nutrient-rich water that often accompanies the sediments. Containment area discharge control from dry dredging projects is less a concern than for wet or hydraulic projects. If the containment area can also serve as the ultimate disposal area, costs are usually greatly reduced. Guidance on containment and disposal is available (USACE, 1987, cited in Cooke et al., 1993a), but the help of experienced professionals is strongly advised.

The productivity of actual dredging depend upon the technique, but a typical dredging year will not involve the removal of more than 100,000 cubic yards (cy) of sediment without multiple pieces of excavation equipment or hydraulic dredges with pumping capacities larger than normal for freshwaters (typically about 1 cy/min). Projects involving 60,000 cy/yr are more typical. Hydraulic or pneumatic dredging is limited to ice-free periods, while other forms of dredging can be conducted anytime.

<u>Monitoring and Maintenance</u>

To assess impacts, biological, chemical and physical monitoring should be performed before, during and after dredging, to document the effectiveness, impacts, and to indicate any changes in water quality. Of particular importance is the monitoring of turbidity or total suspended solids both in the lake and in the discharge water. A monitoring program should be crafted to meet the circumstances of each project.

One of the advantages of dredging is that once the dredging is finished, there is little maintenance required. In cases where there is significant sediment input to the lake, a detention basin or forebay might be constructed to trap the incoming sediments and prolong the benefits of dredging. The detention basin or forebay will have to be cleaned out periodically, and fine sediments will probably still reach the lake. At one site in Wisconsin the sediment traps were filled within 8 years after dredging, indicating that ongoing maintenance was required (Garrison and Ihm, 1991, as cited in Cooke et al., 1993a).

Mitigation

Mitigative measures include partial dredging whereby areas of the lake are dredged while other areas are not. This approach can be used to restore open water while leaving other areas undisturbed. Dredging plans should consider the preservation of fish spawning and nursery areas, waterfowl feeding areas and other sensitive or valuable habitat. In most cases this can be achieved by maintaining some shallow water habitat along the shoreline and in coves.

Several mitigative measures can be used for wet dredging. If bucket dredging is used then a watertight bucket helps reduce the resuspension of sediments. A silt curtain prevents sediments from floating to other areas of the lake (Cooke et al., 1993a). For dry dredging, the restocking of fish and other organisms may be required if migration to and from refuge areas is limited. For reverse layering it is recommended that a boom silt curtain be used to prevent turbidity in areas other than the application site. Additionally, a silt separator may be needed to remove silt from the glacial sand (K-V Associates, Inc., 1991).

A rarely applied but potentially valuable post-dredging mitigative technique is an alum treatment, as it can inactivate any remaining surficial sediment phosphorus and counteract any undesirable inputs from containment area return flow. Of course, alum treatment may be a viable alternative to dredging for inactivating surficial sediment phosphorus, but if dredging is conducted to restore depth or control rooted plants as well as nutrients and algae, phosphorus inactivation can be a valuable final step. It is more likely to be needed with wet or hydraulic dredging than for dry dredging. Careful management of the containment area is important to minimizing such mitigative needs.

3.7.12 Regulations

3.7.12.1 Applicable Statutes

Most dredging projects require multiple permits. A MEPA review is required where applicable thresholds (Appendix II) are exceeded, and will help determine permit needs. If state funds are being used, or if other MEPA thresholds (Appendix II) are exceeded, an Environmental Impact Report may be required. A Notice of Intent must be sent to the Conservation Commission with a copy to the Department of Environmental Protection Regional Office. If the proposed project occurs within an Estimated Habitat of Rare Wildlife in the most recent version of the Natural Heritage Atlas, a copy of the Notice of Intent must be submitted to the Natural Heritage and Endangered Species Program (NHESP) within the MDFG for review (Appendix II). If the proposed project occurs within a Priority Habitat of Rare Species in the most recent version of the Natural Heritage Atlas, the project proponent must submit project plans to the NHESP for an impact determination. An Order of Conditions must be obtained prior to work.

A Chapter 91 permit (Appendix II) is required for dredging and structure installation in Great Ponds. All dredging projects over 100 cubic yards will normally require a Section 401 Water Quality Certificate from the Department of Environmental Protection, and if dredged materials are to be disposed of on land, a Solid Waste Permit is also required from Department of Environmental Protection. There are multiple means for justifying a disposal location, but the most prevalent is a Beneficial Use Determination (BUD), whereby the disposal is categorized as an improvement to the disposal site. If a dam is present and may suffer structural damage or be otherwise altered during drawdown or dredging operations, a permit from the MDCR Office of Dam Safety may be required. If over 100,000 gpd of water is being diverted during the project, a Water Management Act permit may be required through MDEP.

A Section 404 permit from the U.S. Army Corp of Engineers (ACOE) may be required, depending upon the interpretation of this section of the Clean Water Act that prevails at the time of application. In general, any activity associated with dredging that results in filling of federal wetland resources will require a Section 404 permit, but whether or not removal of sediment can be accomplished without any such filling has been the subject of considerable regulatory debate and is still somewhat unsettled. If there is a distinct discharge from the containment area to a surface water, a permit may be required under the National Pollutant Discharge Elimination System (NPDES), administered by the USEPA with input from MDEP.

Depending on site location and scope of work, additional permits and approvals may be required as specified in A-II.1. As it should appear, permitting a dredging project can be a complicated and protracted process, and professional help is strongly advised.

3.7.12.2 Impacts Specific to Wetlands Protection Act

The following overall impact classification is offered as a generalization of impacts, with clarifying notes and caveats as warranted:

1. Protection of public and private water supply – Benefit (water quality improvement); may also affect water quantity by uncapping springs and seepage areas. Short-term limitation on available water is possible during dredging.
2. Protection of groundwater supply – Generally neutral (no significant interaction), although uncapping of springs and seepage areas may increase interaction. Possible adverse impacts below containment area if contaminants leach.
3. Flood control – Generally neutral (no significant interaction), although greater depth could be an asset if drawdown is later practiced for flood control. Possible short-term benefit or detriment during dredging, depending upon technique and flow controls applied.
4. Storm damage prevention – Generally neutral (no significant interaction)), although greater depth could be an asset if drawdown is later practiced for damage control. Possible short-term benefit or detriment during dredging, depending upon technique and flow controls applied.
5. Prevention of pollution – Expected benefit (water quality enhancement), although short-term detriment is possible during unsequestered wet dredging or hydraulic dredging with containment area problems.
6. Protection of land containing shellfish – Possible long-term benefit through water quality enhancement, but potential short-term detriment by direct removal or water quality impacts.
7. Protection of fisheries - Possible long-term benefit through water quality and physical habitat enhancement, but potential short-term detriment by habitat loss during dry dredging or water quality impairment during wet dredging. No major adverse impacts expected from hydraulic dredging.

8. Protection of wildlife habitat – Expected long-term benefit (water quality enhancement, invasive plant control), but possible short-term detriment by habitat loss during dry dredging or water quality impairment during wet dredging. No major adverse impacts expected from hydraulic dredging.

Impacts to interests of the Wetlands Protection Act from a specific dredging project are highly dependent upon site-specific features and project design.

3.7.13 Costs

Because the cost per acre varies depending on the volume of material removed, costs are usually expressed per cubic yard (cy) of material removed. Generally, the larger the project, the smaller the cost per cubic yard, with costs being higher in eastern Massachusetts (C. Carranza, BEC, pers. comm., 1996). The proper way to estimate dredging costs is to consider each element of the project, which may vary dramatically among projects. The total cost can be divided by the total yardage to get a cost per cy, but this may not be especially meaningful in estimating other dredging projects. Nevertheless, a typical range of costs for dredging projects in recent years is $7 to $20/cy, with $10/cy suggested as a rough estimator for considering the general magnitude of a project under initial consideration. It is important, however, to develop a more careful estimate during further project planning.

Dry dredging expenses for several Massachusetts projects illustrate the range of costs. The total project cost for the restoration of Puffers Pond (Factory Hollow Pond) in Amherst, MA in the 1980s was $338,800. Ten feet of sediments were removed across the 9.6-acre pond (74,238 cubic yards) for a cost of $4.56/cy. Included in this cost was the project design, construction of a sediment trap, draining of the pond, and sediment removal (Tighe & Bond, 1994). Over 220,000 cy of sediment were removed from Dunns Pond in Gardner, MA in the early 1980s at a cost of $1,264,000 ($5.74/cy), although a filter berm for cleaning incoming storm water was also included in this cost. About 30,000 cy was removed from Bulloughs and City Hall Ponds in Newton in 1993 at a total cost of about $400,000 ($13.33/cy), although this included additional watershed work and landscaping. Dredging of 15,000 cy from Hills Pond in Arlington, MA in 1995 cost $278,000 ($18.53/cy), including engineering, permitting, dredging and park restoration. The storm water management system to protect Hills Pond was a separate cost. Halls Pond in Brookline could not be dredged affordably, given contamination with benzene compounds; the estimated cost of disposal of those contaminated sediments was in excess of $50/cy.

Total cost can be reduced if the dredged material is clean enough to be sold as a soil amendment. In the case of Lake Trummen, Sweden, the dredged material was sold as topsoil for about $3.43/cy (Cooke et al., 1993a). Lesser revenues were realized from more local projects, including Bantam Lake in CT ($1.00/cy) and Dunns Pond in MA ($0.50/cy), both conducted in the 1980s.

Costs for reverse layering of sediments were estimated at $10,000/acre in 1991 (K-V Associates, Inc., 1991). This technique has not been used enough to provide any general estimate of costs.

3.7.14 Future Research Needs

Research should be continued on the reverse layering of sediments to determine the effectiveness, impacts, and feasibility of implementing this technique. Further investigation of the actual risk from contaminated sediment disposal is needed to determine the appropriate level of protection in dredged sediment disposal. Additional research on long-term impacts of dredging on biota would also be helpful to document the severity of impacts and rate of recovery.

3.7.15 Summary

If properly applied to a shallow lake with significant internal supplies of phosphorus, dredging can produce dramatic improvement in water clarity as well as satisfy the more common goals of increased depth and reduced macrophyte density. In some cases, dredging is the only solution to restoring a pond that is filling in and losing depth. Due to the cost and potential for impacts from some approaches, dredging is usually applied only if less costly or intrusive options are ineffective or infeasible. If applicable and properly applied, dredging can be very effective for the control of nutrients, and can provide control of algae and macrophytes.

Dredging can be accomplished with water still in the lake or with the lake in a dry state. Potential adverse impacts will vary with the method chosen, and the choice of method will depend upon the ability or desire to drain the lake. Dry dredging tends to facilitate the most complete removal of sediment and allows complete physical restructuring of the aquatic habitat, but will impact most lake biota at least temporarily. Excavation with conventional equipment under wet conditions leaves some aquatic habitat during dredging, but will usually create a high level of disturbance in that habitat unless it is somehow sequestered from the active dredging area. Hydraulic or pneumatic dredging minimizes unwanted impacts, but is limited by rocks, stumps and other obstructions, and requires a more sophisticated containment area.

The most significant limitations to sediment removal are sediment quality, finding a suitable location for disposal of the sediments, and the high cost of implementing this technique. The long and potentially difficult process of obtaining all of the permits and acquiring land for sediment disposal should not be underestimated. Contaminated sediments pose additional problems for permitting dredging and can greatly increased costs. Constructing and properly maintaining the containment area is critical to minimizing adverse impacts. Ultimate disposal in the initial containment area can minimize costs, but movement from the dewatering area to a more final disposal area is more common.

3.8 ADDITIONAL TECHNIQUES

Two additional techniques warrant mention here in connection with control of nutrients and associated algal production. Neither has enjoyed substantial application in Massachusetts, but either could be practiced more, has been used in some cases, and may provide benefits. Specific information on each is insufficient to provide a review similar to the other techniques in this section, but future research and application may expand our knowledge of these approaches.

3.8.1 Bacterial Additives

The use of bacterial additives in lakes and ponds has received some attention in recent years, but little detailed scientific study. The theory is simple: add natural or engineered bacteria to the aquatic environment that will out-compete algae for nutrients, binding up the supply of N or P and reducing available concentrations in the lake. In practice, most bacterial additives focus on nitrogen, which would seem to favor blue-green algae that can fix gaseous nitrogen. As nitrogen-fixing blue-greens include some of the most objectionable bloom-forming algae, the value of this approach is unproven. Likewise, it is not clear that a bacterial community capable of precluding algal blooms would not itself constitute an impairment of aquatic conditions. In some cases, practitioners claim bacteria additives consume organic sediments, thus “dredging” the pond, albeit anecdotally with limited supporting data.

3.8.2 Removal of Bottom-Feeding Fish

Biomanipulation to reduce nutrient availability and improve lake transparency includes elimination of fish such as the common carp or bullheads that are bottom browsers. Browsing has been shown to release significant amounts of nutrients to the water column as these fish feed and digest food, and harvesting these fish has resulted in increased clarity in some cases (Baker et al., 1993). It has been suggested that alternative stable equilibria exist for lakes, based on biological structure (Scheffer et al., 1993), and removal of bottom feeding fish could shift the balance. Removing such fish, however desirable, can be very difficult since they tolerate very low levels of dissolved oxygen and high doses of fish poisons. Labor intensive programs appear necessary to achieve substantial reductions in bottom-feeding fish populations (McComas 1993), unless the entire fish population can be sacrificed through complete drawdown, complete freezing, or high doses of rotenone or other fish poisons. A permit to remove any fish species would be required from the MDFG.

3.9 NO MANAGEMENT ALTERNATIVE FOR NUTRIENTS

3.9.1 Overview

The no management alternative for nutrients would exclude all active lake and watershed management programs, but could include monitoring and assessment, and would include normal operation of sewage treatment facilities and other pollution control activities as required by law. As explained in Section 1, the normal tendency for lakes is to gradually accumulate sediments and associated nutrients and to generally become more eutrophic. In consideration of this, the no management alternative would allow lakes to become ever more eutrophic in the future. Eutrophication is expected eventually, even if no human additions of nutrients were involved, but the time scale is greatly reduced by human activities. Most lakes in Massachusetts are influenced by human activities in the watershed, accelerating the eutrophication process in the absence of management. Thus, lack of active lake management will not control eutrophication and can be expected to facilitate acceleration of the process.

The need for management is highly dependent on the ratio of the watershed area to lake area and on the degree of development in the watershed. The predominant natural lake type in Massachusetts is the kettlehole lake, a glacial pothole formed by a stranded block of ice, usually in a sandy outwash plain. Kettlehole lakes have small watershed to lake area ratios, usually <10:1, and great water depth (maximum >30 ft, average >15 ft) relative to lake area (usually <100 acres, although larger ones exist). Water enters naturally as precipitation or groundwater flow, with limited surficial runoff. Human development in the watersheds of kettlehole lakes leads to greater storm water runoff that becomes the primary mode of pollutant entry. As impervious area approaches 10%, water quality impacts are usually detectable (CWP, 2003). As impervious area exceeds 25%, water quality impacts are usually obvious.

There are some natural lakes in Massachusetts formed by natural blockages of stream or river flow, and these normally have watershed to lake area ratios >20:1 and shallow depth (maximum <30 ft, average <15 ft). As the watershed to lake area ratio rises, even natural watershed processes can have an impact on water quality in the lake. At ratios >100:1 it is likely that the lake will become naturally eutrophic in a much shorter time than normally envisioned in the classic lake aging process (lake ontogeny). Where natural processes have caused eutrophication, some support for the no action alternative could be offered. However, all designated uses are unlikely to be fully supported, so even a naturally eutrophic lake might be put on the 303d list.

It should be noted that over half of the lakes in Massachusetts, by area or volume (even excluding Quabbin and Wachusett Reservoirs), exist because of human action (Corbin, ENSR, unpublished data, 1998). Natural processes work to fill in and eutrophy natural lakes over centuries to eons, but many of our created lakes do not have the advantage of great depth or a small watershed to lake area ratio. Quabbin Reservoir, created by human action, has both great depth and a small watershed to lake area ratio, but this is an exception. Small dug ponds may have small watershed to lake area ratios, but will seldom have great depth. Run of the river impoundments will have large watershed to lake area ratios and shallow depth, and are at great risk from accelerated eutrophication. As they were created for human use or as habitat amenities (or both), value is lost if no management occurs.

3.9.2 Effectiveness

The effectiveness of doing nothing to control eutrophication is variable, depending on the condition of the lake and the surrounding watershed. In remote areas with little development, oligotrophic lakes may remain oligotrophic for the foreseeable future. Such lakes are often deep lakes associated with hard bedrock where the weathering rates are low. Nutrient supplies and sedimentation rates are relatively low and such lakes would be expected to remain oligotrophic for a long time in the absence of human influence (Likens, 1972b). However, most Massachusetts lakes are not deep water bodies in isolated areas of forest. Impacts are therefore expected in the vast majority of cases unless management actions are taken. Lakes will not accept elevated levels of nutrients for very long without showing signs of ecological stress and use impairment. The no management alternative is not effective at preventing or controlling eutrophication, and will lead to undesirable conditions in a matter of years to decades, except in the rare case of a low-nutrient lake in an undeveloped watershed.

The impact of doing nothing in lakes that are already eutrophic may not be all that noticeable over a period of several years, and people may adjust their use of the lake accordingly. As habitat for some species diminishes, so will their populations, but other species may take their place until conditions become too severe (e.g., extremely low oxygen, release of toxins from algal blooms). Conditions in the absence of management can indeed get worse, and almost undoubtedly will deteriorate further over a period of years to decades, with high variability in conditions among seasons and years. Where uses have been lost, doing nothing may not have a clearly negative consequence, but the lost opportunity (along with tax revenues and biodiversity) will continue. The no management option in such cases is ineffective at restoring or rehabilitating the lake, but it may not have the obvious negative consequences of no action for a threatened lake that is not yet eutrophic.

3.9.3 Impacts to Non-Target Organisms

In the cases where the no management alternative leads to eutrophication, there can be adverse impacts on a variety of organisms. The most obvious of these occur when the lake reaches a level of eutrophication such that blue-green blooms form and the lake experiences depletion of dissolved oxygen under the ice cover in winter, in the hypolimnion, and/or in areas of dense macrophyte beds during the summer. Such depletion of oxygen can result in fish and invertebrate kills. Dense algal blooms will limit rooted plant cover and diversity. Dense rooted plant growths will affect fish community stability and invertebrate community composition. Highly eutrophic lakes tend to be minimally rich and diverse settings.

In some cases management for nutrient control is incompatible with other management objectives. For example, management for nutrients may reduce plankton and thus reduce fish production (Wagner and Oglesby, 1984). The nature of management focus shifts in accordance with use goals, but the need for management does not abate.

3.9.4 Impacts to Water Quality

If nutrients and sediments are supplied to a lake at high rates due to anthropogenic activities, then water quality will decline. If left uncontrolled, nutrient inputs will result in algal blooms that impact recreation and habitat uses. Water supply use may be impaired by algal blooms that disrupt water treatment and produce toxins. The no action alternative is expected to have adverse impacts on water quality except where the lake is oligotrophic and there is no major loading from the watershed.

3.9.5 Applicability to Saltwater Ponds

The no action alternative is as applicable to Saltwater Ponds as it is to Freshwaters.

3.9.6 Implementation Guidance

3.9.6.1 Key Data Requirements

To determine if the no management alternative has any applicability to a lake and watershed, the lake and watershed condition must be known. Only in rare cases of clean lakes in undeveloped watersheds is this approach usually justifiable. Temporary lack of management may be justified for lakes already in seriously degraded condition, while planning for management proceeds. Funding issues often dictate that no management be taken, but this is not a valid use of this "technique".

3.9.6.2 Factors that Favor this Approach

The following considerations are indicative of appropriate application of no management for reductions in nutrient concentrations and control of algae in lakes:

The lake is in an acceptable condition for designated and desired uses.

There are no apparent threats to lake condition.

Compliance with all federal and state laws relating to pollution control has been achieved.

3.9.6.3 Performance Guidelines

Planning and Implementation

No planning or implementation typically accompanies the no action alternative, although protective action would be warranted where the no action alternative was a valid approach.

Monitoring and Maintenance

No monitoring or maintenance typically accompanies the no action alternative, although data availability is critical to determining if this approach is valid.

Mitigation

No mitigative measures apply to the no management alternative.

3.9.7 Regulations

3.9.7.1 Applicable Statutes

Regulations do not apply directly to the no management alternative. It should be noted, however, that the Commonwealth is required to maintain and monitor water quality as specified under the federal Clean Water Act. In addition, towns are required to close swimming beaches if safe visibility can not be maintained or bacterial standards are exceeded. It is important to note also that no management in these cases runs counter to the USEPA water quality goals of attaining fishable and swimmable water bodies. Action may therefore be mandated by federal or state law, necessitating abandonment of the no management alternative. It should be noted that the no management alternative is generally practiced as a consequence of lack of funds or lack of knowledge, both of which can be substantial hurdles to successful lake management even when mandated by law.

3.9.7.2 Impacts Specific to Wetlands Protection Act

The following overall impact classification is offered as a generalization of impacts, with clarifying notes and caveats as warranted:

1. Protection of public and private water supply – Detriment (water quality deterioration), although impacts may be neutral in rare cases.
2. Protection of groundwater supply – Detriment (if lake interacts with groundwater) or neutral (if no significant interaction).
3. Flood control – Generally neutral (no significant interaction), although water holding capacity may decline over time.
4. Storm damage prevention – Generally neutral (no significant interaction)), although water holding capacity may decline over time.
5. Prevention of pollution – Detriment (water quality deterioration).
6. Protection of land containing shellfish – Detriment (no protection afforded), but impacts may be neutral in some cases.
7. Protection of fisheries - Possible benefit through increased fertility and production, but potential detriment by habitat loss.
8. Protection of wildlife habitat – Detriment (no protection afforded), but impacts may be neutral in some cases.

3.9.8 Costs

Costs do not apply directly to the no management alternative, although there may be costs associated with the impacts to non-target organisms and water quality. For example, additional fish stocking may be required to maintain or replace fish populations due to fish kills. Additional costs may be incurred for additional filtration or other treatment of drinking water supplies when algal blooms form. Such costs are difficult to estimate and would vary on a case by case basis. The reduction in water clarity may also impact real estate values and property tax revenues (Boyle et al., 1997; Jobin, 1997). A study of Maine lakes indicates that this can amount to a loss of millions of dollars when aggregated for an entire lake (Michael et al., 1996).

3.9.9 Future Research Needs

Evaluation of monitoring data for lakes that have not had any focused lake or watershed management would be helpful in underscoring the results of no management. Long-term data sets would be most desirable, spanning a range of at least 20 years and preferably 50 years. Limited data exist that might fulfill this need, but no detailed analysis has been conducted. It is perhaps more critical that long-term monitoring programs be maintained, to provide such baseline data in the future.

3.9.10 Summary

In summary, the no management alternative may be justified in cases where the lake is relatively deep and oligotrophic and with little change anticipated in the watershed. It may also have limited short-term consequences where the lake is already eutrophic. It is most often practiced as a consequence of lack of funding or knowledge of impacts and causative agents. If, however, the lake is shallow and mesotrophic or eutrophic, and there are significant developed lands, agricultural operations or other nutrient sources within the watershed, then the no management alternative for nutrients will not be effective at limiting eutrophication. The trend toward accelerated eutrophication will have adverse impacts on the natural aquatic community and on human uses of the lake. This eutrophication can lead to reduced species richness and diversity, impaired recreational use or water supply, and lowered property values and tax revenues.